WORK IN PROGRESS

Preliminary Draft Recovery Plan for the North Atlantic Population of Loggerhead Turtle (*Caretta caretta*)

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By approving this document, the FWS Regional Director and NMFS Assistant Administrator for Fisheries will certify that the data used in its development represent the best scientific and commercial data available at the time it was written. Copies of all documents reviewed in development of the plan are available in the administrative record located at U.S. Fish and Wildlife Service, 6620 Southpoint Drive South, Suite 310, Jacksonville, Florida 32216; telephone (904) 232-2580.

LITERATURE CITATION SHOULD READ AS FOLLOWS:

National Marine Fisheries Service and U.S. Fish and Wildlife Service. 200_. Recovery Plan for the North Atlantic Population of Loggerhead Turtle (Caretta caretta), Second Revision. National Marine Fisheries Service. Washington, D.C. ___ pp. + appendices.

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ACKNOWLEDGMENTS

The Services gratefully acknowledge the commitment and efforts of the following individuals to the recovery of the North Atlantic population of the loggerhead turtle. Without their assistance and the dynamic discussions and assistance at recovery team meetings, this revision would not have been possible.

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Additional thanks go to the following for their technical assistance and in drafting many sections of this document:
XXX XXX;

To the National Marine Fisheries Service staff:
Susan Pultz, Therese Conant, and Molly Harrison;

and to the U.S. Fish and Wildlife Service staff:
Chuck Underwood, Gloria Bell, and Debby Crouse.

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EXECUTIVE SUMMARY

TO BE DEVELOPED AT A LATER DATE

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LIST OF ACRONYMS AND ABBREVIATIONS

The following standard abbreviations for units of measurement and other scientific/technical acronyms and terms are found throughout this document:

CCL – curved carapace length
CFR – Code of Federal Regulations
ESA – Endangered Species Act
FFWCC – Florida Fish and Wildlife Conservation Commission
FWS – U.S. Fish and Wildlife Service
GDNR – Georgia Department of Natural Resources
HCP – habitat conservation plan
INBS - Index Nesting Beach Survey
NMFS – National Marine Fisheries Service
NCWRC – North Carolina Wildlife Resources Commission
SCL – straight carapace length
SCDNR – South Carolina Department of Natural Resources
STFD – sea turtle fibropapilloma disease
STSSN – Sea Turtle Stranding and Salvage Network
TED – turtle excluder device
TEWG – Turtle Expert Working Group

PREFACE

This revision of the Recovery Plan for the North Atlantic Population of the Loggerhead Turtle adds new and refines existing recovery program activities for the next __ years. The Recovery Plan is composed of four major sections:

1. Introduction: This section acquaints the reader with the loggerhead turtle, its status, the threats it faces, and past and ongoing conservation efforts. It also serves as a review of the biological literature for this species.

2. Recovery: This section describes the goal of the plan; presents delisting criteria based upon the five listing/recovery factors and population benchmarks to assist in evaluating the status; and objectives, strategy, and actions needed to achieve recovery. The recovery actions are presented in a narrative outline, organized by ___ major objectives: (1) ________; (2) ____________; (3) ____________; and (4) ____________.

3. Implementation Schedule: This section presents the recovery actions from the narrative outline in table format; assigns priorities to the recovery actions; estimates the time necessary to complete the recovery actions; identifies parties with authority, responsibility, or expressed interest in implementation of the recovery actions; and estimates the cost of the recovery actions and recovery program.

4. Appendices: This section presents additional information utilized by the Services and the Recovery Team to draft this revision.

PART I. INTRODUCTION

BACKGROUND

The Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq*.) (ESA), establishes policies and procedures for identifying, listing, and protecting species of wildlife that are endangered or threatened with extinction. The purposes of the ESA are "to provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved, [and] to provide a program for the conservation of such endangered species and threatened species..." The ESA defines an "endangered species" as "any species which is in danger of extinction throughout all or a significant portion of its range." A "threatened species" is defined as "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." The loggerhead sea turtle (*Caretta caretta*) was listed as threatened throughout its range on July 28, 1978 (FWS 1978; 43 FR 32800) and has received Federal protection under the ESA since that time.

The Secretaries of the Department of the Interior and the Department of Commerce are responsible for administering the ESA's provisions as they apply to the loggerhead sea turtle. Day-to-day management authority for endangered and threatened species under the Departments' jurisdictions has been delegated to the U.S. Fish and Wildlife Service (FWS) and the National Marine Fisheries Service (NMFS), respectively. FWS and NMFS share Federal jurisdiction for sea turtles, with FWS having lead responsibility on the nesting beaches and NMFS in the marine environment.

To help identify and guide species recovery needs, section 4(f) of the ESA directs the Secretary to develop and implement recovery plans for listed species or populations. Such plans are to include: (1) a description of site-specific management actions necessary to conserve the species or populations; (2) objective, measurable criteria which, when met, will allow the species or populations to be removed from the endangered and threatened species list; and (3) estimates of the time and funding required to achieve the plan's goals and intermediate steps. Section 4 of the ESA and regulations (50 CFR Part 424) promulgated to implement its listing provisions, also set forth the procedures for reclassifying and delisting species on the Federal lists. A species can be delisted if the Secretary of the Interior and/or the Secretary of Commerce determines that the species no longer meets the endangered or threatened status based upon these five factors listed in Section 4(a)(1) of the ESA:

(1) the present or threatened destruction, modification, or curtailment of its habitat or range;
(2) overutilization for commercial, recreational, scientific, or educational purposes;
(3) disease or predation;
(4) the inadequacy of existing regulatory mechanisms; and
(5) other natural or manmade factors affecting its continued existence.

Further, a species may be delisted, according to 50 CFR Part 424.11(d), if the best scientific and commercial data available substantiate that the species or population is neither endangered nor threatened for one of the following reasons: (1) extinction, (2) recovery, or (3) original data for classification of the species were in error.

NMFS approved the initial recovery plan for the loggerhead sea turtle on September 19, 1984. This initial plan was a multi-species plan for all six species of sea turtles occurring in the U.S. On December 26, 1991, NMFS and FWS approved a separate recovery plan for the U.S. population of the loggerhead sea turtle. In 2001, NMFS and FWS initiated the process to revise the plan for a second time. An Atlantic Loggerhead Sea Turtle Recovery Team (see Acknowledgment Section), consisting of species experts, was established to draft this revision.

Since approval of the first revised plan in 1991, significant research has been accomplished and important conservation and recovery activities have been undertaken. As a result, we have a greater knowledge of the species and its status. This second revision of the recovery plan for the Atlantic loggerhead addresses current threats and needs, highlights conservation accomplishments that have been undertaken since the species was listed, and specifically addresses the planning requirements of the ESA.

OVERVIEW

Loggerhead sea turtles nest within the continental United States from Texas to Virginia. Major nesting concentrations in the U.S. are found on the coastal islands of North Carolina, South Carolina, and Georgia, and on the Atlantic and Gulf coasts of Florida (Hopkins and Richardson 1984) (Figure 1). Within the Atlantic Ocean basin, loggerheads also nest in Mexico and the Caribbean.

Figure 1. Annual loggerhead nesting estimates from the continental U.S., Mexico, and the Caribbean, 2002.

From a global perspective, the U.S. nesting aggregation is of paramount importance to the survival of the species and is second in size only to that which nests on islands in the Arabian Sea off Oman (Ross 1982, Ehrhart 1989). The status of the Oman colony has not been evaluated recently, but its location in a part of the world that is vulnerable to disruptive events (e.g., political upheavals, wars, catastrophic oil spills) is cause for considerable concern (Meylan *et al*. 1995). The loggerhead nesting aggregations in Oman, the U.S., and Australia account for about 88 percent of nesting worldwide (National Marine Fisheries Service and U.S. Fish and Wildlife Service 1991).

The major threats faced by loggerheads include incidental take from commercial fishing operations and channel dredging; loss or degradation of nesting habitat from coastal development and beach armoring; disorientation of hatchlings by beachfront lighting; excessive nest predation by native and non-native predators; degradation of foraging habitat; marine pollution and debris; watercraft strikes; and disease.

The highly migratory behavior of loggerheads makes them shared resources among many nations. Therefore, conservation efforts for loggerhead populations in one country may be jeopardized by activities in another. Protecting loggerhead sea turtles on U.S. nesting beaches and in U.S. waters alone, therefore, is not sufficient to ensure the continued existence of the species. However, sea turtle protection programs in many countries are not well organized or supported and, in this context, protection of the U.S. loggerhead population takes on international significance. Although this revised recovery plan focuses on activities to recover the loggerhead in the U.S., it also recognizes and encourages cooperative efforts with other nations to ensure the survival of the species.

A. TAXONOMY

UNDER DEVELOPMENT

B. DESCRIPTION (MORPHOLOGICAL)

UNDER DEVELOPMENT

C. POPULATION DISTRIBUTION

The loggerhead sea turtle occurs throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans. However, the majority of loggerhead nesting is at the western rims of the Atlantic and Indian oceans (Figure 1). The most recent reviews show that only two loggerhead nesting beaches have greater than 10,000 females nesting per year: South Florida (U.S.) and Masirah (Oman). Those beaches with 1,000 to 9,999 females nesting each year are North Florida through North Carolina (U.S.), Cape Verde Islands (Spain, eastern Atlantic off Africa), and Western Australia (Australia) (Bolten and Witherington in press). Smaller nesting aggregations with 100 to 999 nesting females annually occur in Northwest Florida (U.S.), Cay Sal Bank (Bahamas), Quintana Roo and Yucatán (Mexico), Sergipe and Northern Bahia (Brazil), Southern Bahia to Rio de Janerio (Brazil), Tongaland (South Africa), Mozambique, Arabian Sea Coast (Oman), Halaniyat Islands (Oman), Cyprus, Peloponnesus (Greece), Island of Zakynthos (Greece), Turkey, and Queensland (Australia).

Although the major nesting concentrations in the United States are found in South Florida, loggerheads nest from Texas to Virginia. Total estimated nesting in the U.S. is approximately 68,000 to 90,000 nests per year. About 80 percent of loggerhead nesting in the southeastern U.S. occurs in six Florida counties (Brevard, Indian River, St. Lucie, Martin, Palm Beach, and Broward Counties). Adult loggerheads are known to make considerable migrations between foraging areas and nesting beaches. During non-nesting years, adult females from U.S. beaches are distributed in waters off the eastern U.S. and throughout the Gulf of Mexico, Bahamas, Greater Antilles, and Yucatán.

Genetic research involving analysis of mitochondrial DNA has identified four different loggerhead nesting subpopulations in the southeastern United States: (1) the Northern Subpopulation occurring from North Carolina through Northeast Florida; (2) South Florida Subpopulation occurring from just north of Cape Canaveral on Florida's east coast and extending up to around Sarasota on Florida's west coast; (3) Dry Tortugas, Florida, Subpopulation, and (4) Northwest Florida Subpopulation occurring on Florida's Panhandle beaches (Bowen *et al*. 1993; Bowen 1994, 1995; Encalada *et al*. 1998; Pearce 2001). These data indicate that gene flow between these four regions is very low. If nesting females are extirpated from one of these regions, regional dispersal will not be sufficient to replenish the depleted nesting subpopulation.

D. STATUS (POPULATION SIZE AND TRENDS)

The Recovery Team decided to divide Atlantic loggerheads into Recovery Units. Recovery Units are geographic or otherwise identifiable subunits of a listed entity that "individually" are "necessary" to conserve genetic robustness, demographic robustness, important life history stages, or some other feature necessary for long-term sustainability of the entire listed entity. Recovery Units are not necessarily self-sustaining viable units on their own, but instead need to be collectively recovered to ensure recovery of the entire listed entity.

Five Recovery Units for the loggerhead have been identified based on mitochondrial DNA data that have identified different nesting assemblages. The first four Recovery Units coincide with the four genetically different nesting assemblages that have been identified in the southeastern United States. The fifth Recovery Unit is a combination of all other nesting assemblages of turtles that nest outside the U.S. but occur within U.S. waters during some portion of their lives. This will ensure that all turtles occurring in U.S. waters are included within a Recovery Unit, even those that originated on nesting beaches outside the U.S.

The five Recovery Units are: (1) Northern Recovery Unit, which includes Northeast Florida through North Carolina, (2) South Florida Recovery Unit, (3) Dry Tortugas Recovery Unit, (4) Florida Panhandle Recovery Unit, and (5) "Other" (Yucatán, Cuba, Bahamas, Brazil, Cape Verde, and Mediterranean) Recovery Unit. Each Recovery Unit will have distinct recovery criteria, and each recovery criterion must be met to ensure recovery of the species.

D.1. NORTHERN RECOVERY UNIT

The Northern Recovery Unit is the second largest loggerhead nesting aggregation in the western north Atlantic. Annual nest totals from northern beaches ranged from 3,629 to 6,642 between

1989 and 1998, representing approximately 1,287 nesting females per year (adapted from TEWG 2000 – the Northeast Florida data reported by TEWG 2000 was revised to include only those nesting activities reported from Amelia Island north to the Florida-Georgia border). In this summary, loggerhead nest counts from standardized daily beach surveys and aerial surveys conducted by South Carolina Department of Natural Resources were used to assess population trends for Northern Recovery Unit beaches.

Standardized Nest Surveys

Historically, survey effort on Northern Recovery Unit beaches has been variable making it difficult to assess nesting trends. In order to standardize the data used in our analysis, we included only annual nest totals from beaches that met the following criteria: (1) surveys began between May 1-May 15 and continued through the end of the nesting season (August 31), (2) surveys were conducted daily although 1-2 days may have been missed due to logistical difficulties, and (3) survey area was standardized throughout the length of the study (± 1 kilometer for changes in beach profile).

Two time-series were examined for trends, 30 and 21 years. For the 30-year time-series, only three beaches (Wassaw, Blackbeard, and Little Cumberland) met the criteria mentioned above for analysis. Figure 2 shows the summed nest counts from these three beaches from 1973-2002. A regression of log-transformed nest totals showed a significant (P=0.0363) annual decrease of 1.2 percent in loggerhead nesting. However, this trend must be viewed with caution because the datasets used for this analysis represented only 6 percent of the Northern Recovery Unit nesting and may not be representative of overall population trends.

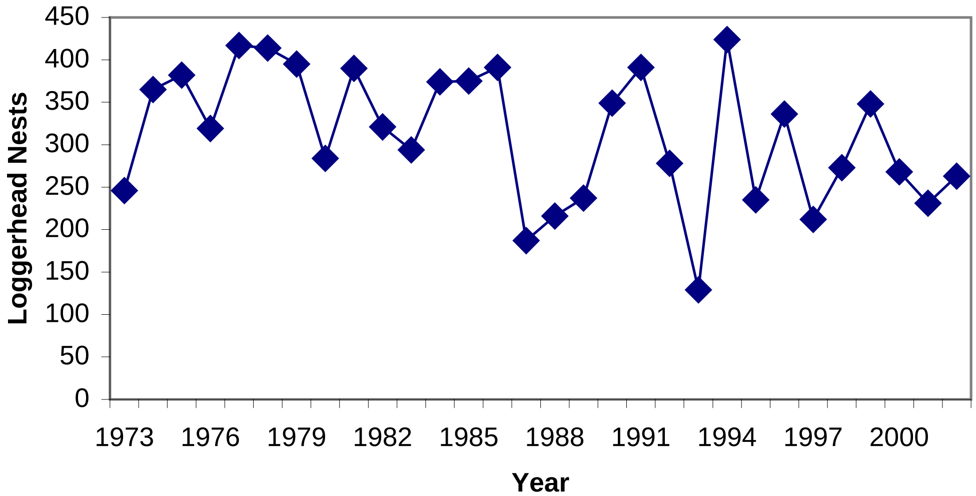

Figure 2. Summed annual loggerhead nest counts from three Northern Recovery Unit beaches (Wassaw, Blackbeard, and Little Cumberland), 1973-2002.

In order to gain a more representative sample, we examined a shorter time-series (21 years). Figure 3 shows the summed nest totals from eight Northern Recovery Unit beaches (South, Cape, Edisto Beach State Park, Fripp, Wassaw, Ossabaw, Blackbeard, and Little Cumberland) from 1982-2002. These totals represent approximately 31 percent of Northern Recovery Unit annual nesting. A regression of log-transformed nest totals showed no trend (P=0.16). According to power analysis conducted using the TRENDS program (Gerrodette 1993), 29 years of data is required to detect a 2.0 percent annual change in nests (alpha=.20, power=.90). By using a shorter time-series, we obtain a more representative sample of the population, but lose power to detect nesting trends.

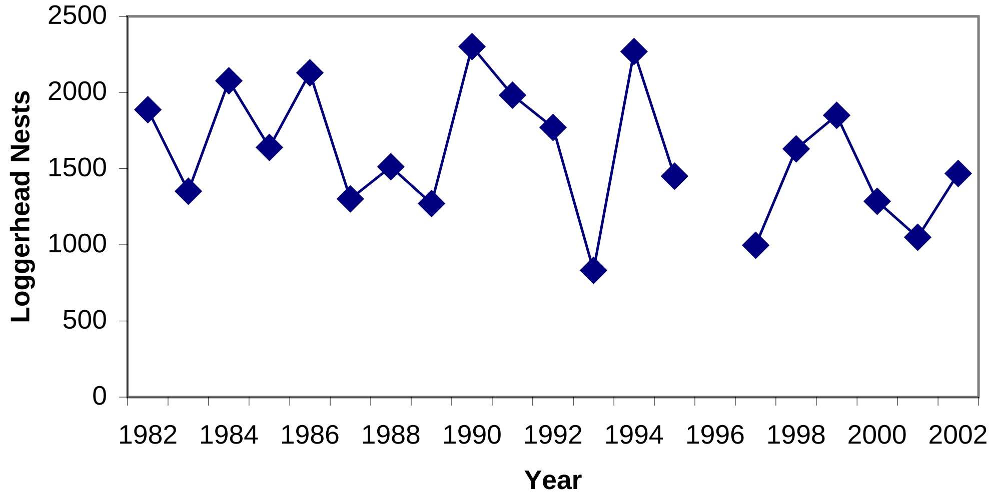

Figure 3. Summed annual loggerhead nest counts from eight Northern Recovery Unit beaches (South, Cape, Edisto Beach State Park, Fripp, Wassaw, Ossabaw, Blackbeard, and Little Cumberland), 1982-2002.

In an attempt to utilize all available nesting data, we also used a statistical technique called Route Regression to assess nesting trends on Northern Recovery Unit beaches. Route Regression was developed by Geissler and Sauer (1990) to assess regional trends in breeding bird counts. A regional trend is calculated as a weighted average of the individual route (beach) trends. The dataset was analyzed using the program ESTEQN (Collins 1997). This technique was attractive because we could use all available data including datasets with missing years. The route-regression analysis showed no significant trend in loggerhead nesting (P=0.55) over the last 38 years. Future work will include power analysis to estimate the percent annual change detectable using this technique.

Aerial Nest Surveys
Standardized aerial nest surveys conducted by the South Carolina Department of Natural Resources represent another dataset for assessing Northern Recovery Unit nesting trends. Beginning in 1980, loggerhead nests were surveyed from the side window of a fixed wing aircraft. Twelve surveys were conducted biweekly during June and July each year for the entire South Carolina coast with the exception of Myrtle Beach. Surveys were conducted for three consecutive years followed by two years of “spot check” surveys. An annual nest total was derived by estimating the percent nesting represented by the 12 flight days (after adjusting for bias from ground truth beaches) and then extrapolating out to an overall total for the season (from a summary composite curve of nesting). Figure 4 shows loggerhead nest estimates from South Carolina aerial surveys, 1980-2002. A regression of log-transformed nest totals showed a

significant (P=0.0009) annual decrease of 3.1 percent in loggerhead nesting. The South Carolina data represents approximately 59 percent of Northern Recovery Unit nesting totals.

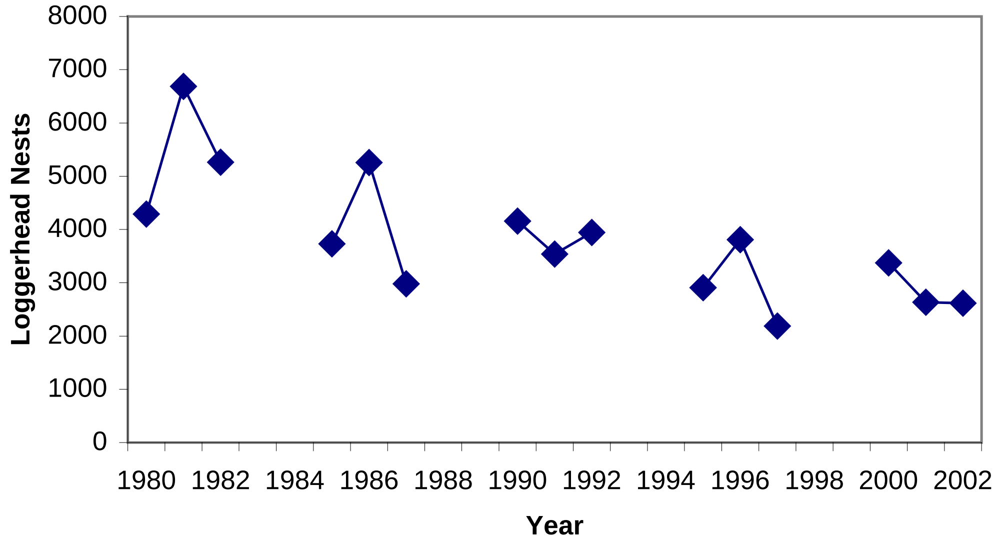

Figure 4. Loggerhead nest estimates for South Carolina from aerial surveys, 1980-2002.

Summary
Estimates of loggerhead nesting trends from standardized daily beach surveys ranged from no trend to a 1.2 percent annual decrease in nesting. Because of annual variability in loggerhead nesting, 28 to 30 years of standardized survey data are necessary to detect $\pm$2 percent annual change in nesting. Nest totals from aerial surveys conducted by South Carolina Department of Natural Resources showed a 3.1 percent annual decline in nesting since 1980.

D.2. SOUTH FLORIDA RECOVERY UNIT

The South Florida Recovery Unit is the largest loggerhead nesting assemblage in the Atlantic. A near census of the South Florida Recovery Unit undertaken from 1998 to 2002 reveals a mean of 75,459 nests per year, which equates to about 18,405 females nesting per year (Florida Fish and Wildlife Conservation Commission (FFWCC), unpublished data). This near census provides the best estimate of total abundance but because of variable survey effort, these numbers cannot be used to assess trends.

Data from all beaches where nesting activity has been recorded indicate that the South Florida Recovery Unit has shown significant increases over the last 25 years. However, an analysis of nesting data from the Florida Index Nesting Beach Survey (INBS) Program from 1989 to 2002, a period encompassing index surveys that are more consistent and more accurate than surveys in previous years, has shown no detectable trend and, more recently (1998 through 2002), has shown evidence of a declining trend (Blair Witherington, Florida Fish and Wildlife Conservation Commission, personal communication, 2002) (Figure 5). Given inherent annual fluctuations in nesting and the short time period over which the decline has been noted, caution is warranted in interpreting the decrease in terms of nesting trends.

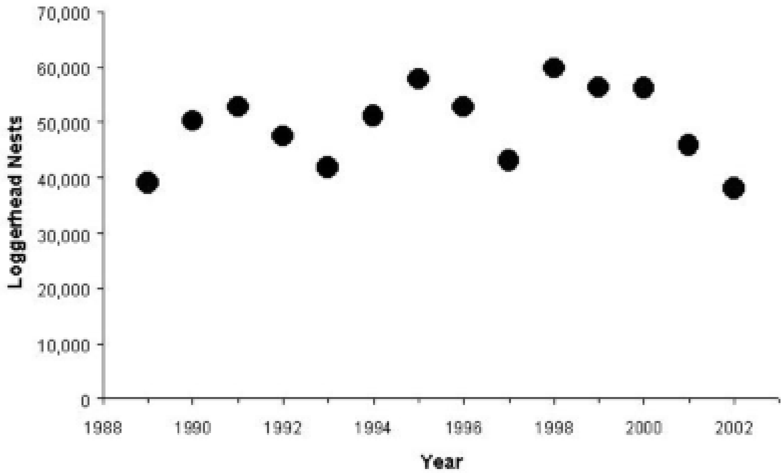

Figure 5. Annual loggerhead nests counted at the South Florida index nesting beaches, 1989-2002.

D.3. FLORIDA PANHANDLE RECOVERY UNIT

The Florida Panhandle Recovery Unit appears to be the third largest in size. A near census of the Florida Panhandle Recovery Unit undertaken from 1989 to 2002 reveals a mean of 1,028 nests per year, which equates to about 251 females nesting per year (FFWCC, unpublished data).

Evaluation of long-term nesting trends for the Florida Panhandle subpopulation is difficult because of changed and expanded beach coverage. However, there are 6 years of INBS data for the Florida Panhandle Recovery Unit, but the time series is too short to detect a trend (Blair Witherington, FFWCC, personal communication, 2003) (Figure 6).

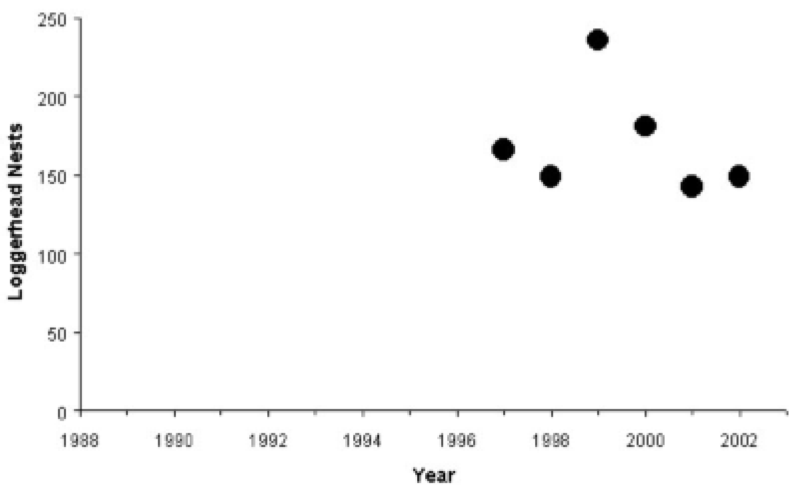

Figure 6. Annual loggerhead nests counted at the Florida Panhandle index nesting beaches, 1997-2002.

D.4. DRY TORTUGAS RECOVERY UNIT

The Dry Tortugas Recovery Unit, located west of the Florida Keys, is the smallest of the identified Recovery Units. A near census of the Dry Tortugas Recovery Unit undertaken from 1995 to 2001 reveals a mean of 213 nests per year, which equates to about 50 females nesting per year (FFWCC, unpublished data).

The trend data for the Dry Tortugas Recovery Unit are from beaches that are not part of the INBS program but have moderately good monitoring consistency. There are 7 years of data for this Recovery Unit, but the time series is too short to detect a trend (Blair Witherington, FFWCC, personal communication, 2003) (Figure 7).

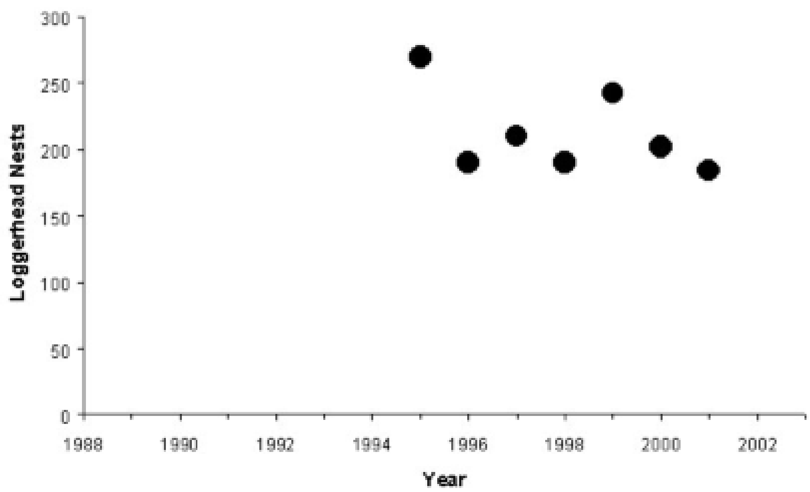

Figure 7. Annual loggerhead nests counted at the Dry Tortugas nesting beaches, 1995-2001.

D.5. "OTHER" RECOVERY UNIT

The "Other" Recovery Unit is a combination of all other nesting assemblages of loggerheads that nest outside the U.S. but occur within U.S. waters during some portion of their lives. This will ensure that all loggerheads occurring in U.S. Atlantic Ocean basin waters are included within a Recovery Unit, even those that originated on nesting beaches outside the U.S. This Recovery Unit includes loggerheads that nest in the Yucatán, Cuba, Bahamas, Brazil, Cape Verde, and Mediterranean.

One of the nesting assemblages in this Recovery Unit occurs in the Yucatán Peninsula of Mexico. The Yucatán nesting assemblage had 1,052 nests in 1998 and has a nesting trend believed to be stable or increasing, but with little nesting survey data (Turtle Expert Working Group 2000).

E. BIOLOGICAL CHARACTERISTICS -- OVERVIEW

The generalized life history of Atlantic loggerhead sea turtles is shown in Figure 8. The three basic ecosystems in which loggerheads live are the:

- Terrestrial zone (supralittoral) - the nesting beach where both oviposition and embryonic development occur.
- Neritic zone - the inshore marine environment (from the surface to the sea floor) where water depths do not exceed 200 meters. The neritic zone generally includes the continental shelf, but in areas where the continental shelf is very narrow or nonexistent, the neritic zone conventionally extends to areas where water depths are less than 200 meters.
- Oceanic zone - the vast open ocean environment (from the surface to the sea floor) where water depths are greater than 200 meters.

Within the two marine ecosystems:

- Organisms are pelagic if they occupy the water column, but not the sea floor, in either the neritic zone or oceanic zone. Organisms are epipelagic if they occupy the upper 200 meters in the oceanic zone.
- Organisms on the sea floor in either the neritic zone or oceanic zone are described as benthic or demersal.

Bolten (in press a, b) reviews this terminology with respect to sea turtle life history; see Lalli and Parsons (1993) for review of basic oceanographic terminology.

The life history stages are described in the following sections.

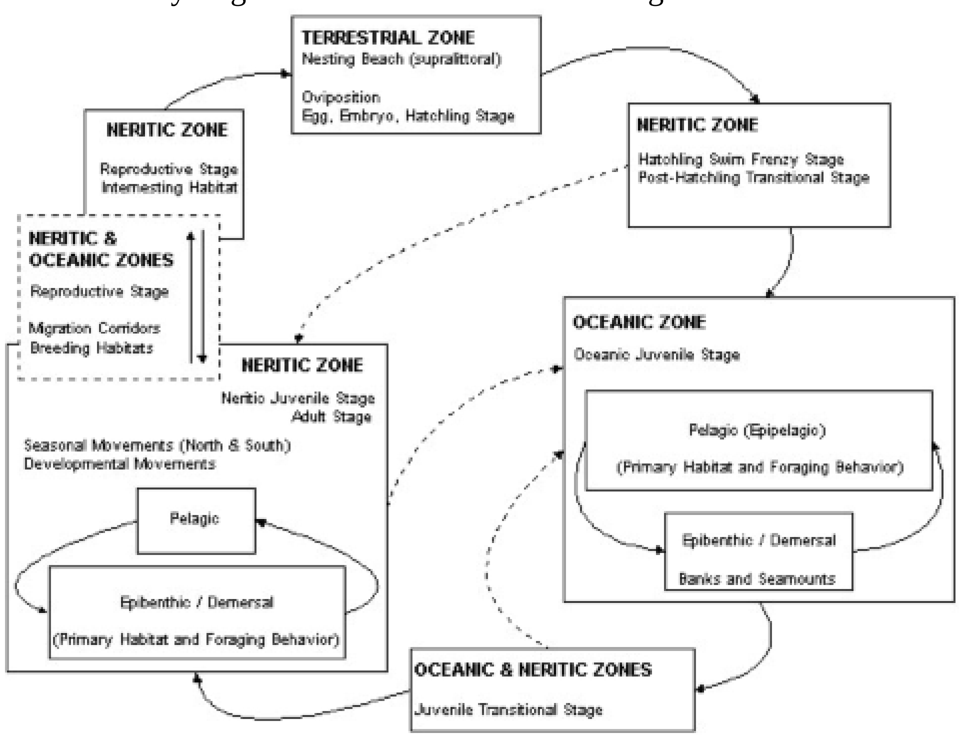

Figure 8. Generalized life history of Atlantic loggerhead sea turtles (from Bolten in press).

E.1. TERRESTRIAL ZONE (NESTING BEACH)

Nesting Habitat

Loggerheads nest on ocean beaches and occasionally on estuarine shorelines with suitable sand. Nests are typically made between the high tide line and the dune front (Routa 1968, Witherington 1986, Hailman and Elowson 1992). Wood and Bjorndal (2000) evaluated four environmental factors (slope, temperature, moisture, and salinity) and found that slope had the greatest influence on nest-site selection in loggerheads. Loggerheads appear to prefer relatively narrow, steeply sloped, coarse-grained beaches, although nearshore contours may also play a role in nesting beach site selection (Provancha and Ehrhart 1987). Kikukawa *et al*. (1999) found that sand softness and distance from the nearest human population center were the best predictors of loggerhead nesting density among 23 factors measured on 300 nesting beaches in Japan.

Nest Characteristics/Requirements

Sea turtle eggs require a low-salinity, high-humidity, well-ventilated substrate for development (Miller 1997, Miller *et al*. in press). Mean clutch size varies from about 100 to 126 eggs along the southeastern United States coast (Dodd 1988). Loggerhead nests incubate for variable periods of time. The length of the incubation period (commonly measured from the time of egg deposition to hatchling emergence) is inversely related to nest temperature, such that between 26^{o}C and 32^{o}C, a change of 1^{o}C adds or subtracts approximately 5 days (Mrosovsky 1980).

The incubation period for loggerhead nests ranges from about 43 to 65 days along the central and south Florida Atlantic coast (Davis and Whiting 1977, Ehrhart and Witherington 1987, Bain *et al*. 1997), 43 to 76 days along the south Florida Gulf coast (Jerris Foote, Mote Marine Laboratory, personal communication, 2002), 44 to 84 days for the north Florida Atlantic coast (Ecological Associates, Inc. 2002), 45 to 95 days for the Florida panhandle and Alabama (Moody *et al*. 2000, Celeste South, FWS, personal communication, 2002), 50 to 76 days in Georgia (Kraemer 1979; Dodd and Mackinnon 1999, 2000, 2001), 49 to 62 days in South Carolina (Caldwell 1959), and 56 to 75 days in North Carolina (Crouse 1985, Ferris 1986). However, incubation periods as long as 102 days have been documented in North Carolina (Crouse 1985; North Carolina Wildlife Resources Commission (NCWRC), unpublished data).

The warmer the sand surrounding the egg chamber, the faster the embryos develop (Mrosovsky and Yntema 1980). Sediment temperatures prevailing during the middle third of the incubation period also determine the sex of hatchling sea turtles (Mrosovsky and Yntema 1980). Highest incubation temperatures produce nearly all female hatchlings, and lowest incubation temperatures produce nearly all males. The pivotal temperature (i.e., the incubation temperature that produces equal numbers of males and females) for hatchling sex ratio in loggerheads is approximately 29^{o}C (Limpus *et al*. 1983, Mrosovsky 1988, Marcovaldi *et al*. 1997). However, clutches with the same average temperature may result in different sex ratios depending on the fluctuation of temperature during incubation (Georges *et al*. 1994). Moisture conditions in the nest similarly influence incubation period, hatching success, and hatchling size (McGehee 1990, Carthy *et al*. in press).

Hatchling Emergence Behavior

Loggerhead hatchlings pip and escape from their eggs over a 1- to 3-day interval and move upward and out of the nest over a 2- to 4-day interval (Christens 1990). The time from pipping to emergence ranges from 4 to 7 days with an average of 4.1 days (Godfrey and Mrosovsky 1997). Hatchlings emerge from their nests en masse almost exclusively at night, and presumably using decreasing sand temperature as a cue (Hendrickson 1958, Mrosovsky 1968, Witherington *et al*. 1990). Moran *et al*. (1999) concluded that a lowering of sand temperatures below a critical threshold, which most typically occurs after nightfall, is the most probable trigger for hatchling emergence from a nest. After an initial emergence, there may be secondary emergences on subsequent nights (Carr and Ogren 1960, Witherington 1986, Ernest and Martin 1993).

Emergence marks the beginning of a period of high levels of activity (the hatchling frenzy) during which hatchlings enter the sea and swim away from land (Wyneken and Salmon 1992). Hatchlings use a progression of orientation cues to guide their movement from the nest to oceanic environments where they spend their early years (Lohmann and Lohmann in press). Hatchlings first use light cues to find the ocean. On naturally lighted beaches without artificial lighting, ambient light from the open sky and reflected off the ocean creates a relatively bright horizon compared to the dark silhouette of the dune and vegetation landward of the nest. This contrast guides the hatchlings to the ocean (Daniel and Smith 1947, Limpus 1971, Salmon *et al*. 1992, Witherington 1997, Witherington and Martin 2000). Upon entering the surf, hatchlings swim incessantly in an offshore direction for about 20 hours (Wyneken and Salmon 1992). Wave direction and magnetic fields are thought to be responsible for leading the hatchlings to offshore habitats (Salmon and Lohmann 1989, Lohmann *et al*. 1990, Wyneken *et al*. 1990, Lohmann 1991, Wyneken and Salmon 1992, Light *et al*. 1993, Lohmann and Lohmann 1994, Lohmann and Lohmann in press) where they spend their prolonged oceanic life stage (Carr 1986, 1987; Bolten in press).

E.2. Neritic Zone: Hatchling Swim Frenzy Stage and Post-hatchling Transitional Stage

Swim Frenzy

Immediately after hatchlings emerge from the nest, they begin a period of frenzied activity. During this active period, hatchlings move from their nest to the surf, swim and are swept through the surf zone, and continue swimming incessantly away from land for approximately 20 to 30 hours (Carr and Ogren 1960, Carr 1962, Carr 1982, Wyneken and Salmon 1992, Witherington 1995). Orientation and movement of hatchlings from nest to sea is described in the preceding section. Upon reaching the swash zone, hatchlings may be tumbled back up the beach by the swash of several waves before they are able to enter the swash just as it slows and then backswashes into the surf. As hatchlings are buoyed by water, they begin swimming in a pattern of subsurface powerstroking interrupted by brief surface dogpaddling/breathing (Salmon and Wyneken 1987, Witherington 1995). In the surf, loggerhead hatchlings, like green turtle hatchlings (Carr 1962), make powerstroking dives as breaking waves approach so that they are swept seaward by wave motion near the bottom. Seaward of the breakers, hatchlings orient away from land and continue swimming with a pattern of 5- to 20-second bouts of subsurface powerstroking interrupted by 1- to 3-second bouts of surface dogpaddling/breathing

(Witherington 1995). In still water, loggerhead hatchlings travel approximately 1.0 to 1.4 kilometers per hour (Salmon and Wyneken 1987, Witherington 1995). Mortality from fish predation was 5 of 74 hatchlings swimming within approximately 1 kilometer from the beach and within 2 hours of entering the Atlantic at three Florida locations (Witherington and Salmon 1992). Predation over reefs and at hatcheries may be higher (Wyneken and Salmon 1996, Wyneken *et al*. 1998).

Orientation cues used by hatchlings as they crawl, swim through the surf, and migrate offshore are discussed in detail by Lohmann and Lohmann (in press). Hatchlings that have just entered the water use both the surge motion and orbital movement of water to guide themselves seaward (Wyneken *et al*. 1990, Lohmann *et al*. 1995, Wang *et al*. 1998). Initial orientation of hatchlings in the water is into oncoming waves with other cues having little or no effect on orientation when waves are present. Wave direction near the beach correlates with offshore direction because waves that reach the shallow waters near shore refract until they are nearly parallel with the coastline. Farther from land, wave direction is a less reliable indicator of offshore direction. Loggerhead hatchlings tracked in the Atlantic from a Florida beach were observed to continue their offshore headings even as wave direction varied greatly (Witherington 1995). Sometime after an offshore course is established, wave orientation is supplanted by orientation guided by a magnetic compass (Lohmann 1991, Light *et al*. 1993, Lohmann and Lohmann 1994a). Experiments have shown that this magnetic compass is an inclination compass rather than a polarity compass and that its function is similar to the magnetic compass shown by birds (Light *et al*. 1993). Upon first emerging from the nest, loggerhead hatchlings do not show a preference for a magnetic orientation swimming direction and must acquire a directional preference as they crawl and swim from their nest site. The experience of maintaining a directional course, and the effect of light cues and wave cues on crawling and swimming direction, can influence subsequent magnetic orientation (Lohmann and Lohmann 1994a, Goff *et al*. 1998).

Observations of loggerheads swimming in a laboratory setting (Wyneken and Salmon 1992) and at sea (Witherington 1995) reveal a pronounced reduction in activity after 20 to 30 hours post-emergence, although oriented swimming continues for several days afterward. Five hatchlings released near midnight and observed constantly for 2 days during their movement from a Florida beach into the Atlantic swam with the powerstroke and dogpaddle pattern during the first 30 hours. After 30 hours (daylight of their second day at sea), they began to use a lower-energy rear-flipper kick-swimming pattern in addition to powerstroking. On the following night (42 hours post release), hatchlings became inactive and were commonly in a tuck position wherein flippers are held close to the body. Two hatchlings followed more than 3 days continued a diurnal swimming pattern like that observed in hatchlings swimming under natural photoperiods in the laboratory (Wyneken and Salmon 1992). In addition to swimming, hatchlings may stop briefly to move within the floating seaweeds in the genus *Sargassum* located in their path, assume a tuck position in response to subsurface predators, and make dives to 3 meters in response to aerial predators (Witherington and Salmon 1992, Witherington 1995). Hatchlings swimming from land rely on an approximately 5-day store of energy and nutrients within their retained yolk sac (Kraemer and Bennett 1981).

Post-hatchling Transition
Neonate loggerheads that have migrated away from land are commonly found near the Gulf Stream off Florida (Witherington 2002). These post-hatchlings differ from hatchlings during their swim frenzy in that they participate only infrequently in low-energy swimming and that they have begun to feed, relying no longer on their retained yolk (Witherington 2002). As post-hatchlings, loggerheads are pelagic and are best known from neritic waters along the continental shelf. This neritic post-hatchling stage is weeks or months long (Witherington 2002) and may be a transition to the oceanic stage that loggerheads enter as they grow and are carried within ocean currents (Bolten in press).

Post-hatchling loggerheads inhabit areas where surface waters converge to form local downwellings (Witherington 2002). These areas are characterized by linear accumulations of floating material, especially *Sargassum*, and are common between the Gulf Stream and the Southeast U.S. coast, and between the Loop Current and the Florida coast in the Gulf of Mexico. Post-hatchlings within this habitat are observed to be low-energy float-and-wait foragers that feed on a wide variety of floating items (Witherington 2002). Witherington (2002) found that small animals commonly associated with the *Sargassum* community, such as hydroids and copepods, were most commonly found in esophageal lavage samples.

As post-hatchlings, loggerheads may linger for months in waters just off the nesting beach or become transported by ocean currents within the Gulf of Mexico and North Atlantic. Work by Lohmann and Lohmann (1994b, 1996) and Lohmann *et al*. (1999) suggest that loggerheads may continue some oriented swimming in order to keep from being swept into cold North Atlantic currents. Loggerheads have shown an ability to detect their location within currents using magnetic inclination angles and magnetic field strength, and to orient toward the center of the North Atlantic Gyre (Figure 9).

[INSERT FIGURE 9 - MAP OF NORTH ATLANTIC GYRE SYSTEM]

Figure 9. North Atlantic Gyre system.

E.3. OCEANIC ZONE: JUVENILE STAGE

The biology of the oceanic juvenile stage (which will be referred to as the oceanic stage) has recently been reviewed by Bolten (in press a, b); information for this section has been modified from those publications.

Habitat Description
The oceanic juvenile stage begins when the turtles enter the oceanic zone and has been primarily studied in the waters around the Azores and Madeira (Bolten in press a). Other populations exist (e.g., in the region of the Grand Banks off Newfoundland), but data on these populations are very limited. Turtle movements in this stage are both active and passive relative to surface and sub-surface oceanic currents and winds; turtles may use bathymetic features for orientation based on satellite telemetry and remote sensing studies (Bolten, Riewald, and Bjorndal, unpublished data). These turtles are epipelagic, spending 75 percent of their time in the top 5 meters of the water

column, 80 percent of the dives are between 2 to 5 meters with the remainder of the dives distributed throughout the top 100 meters of the water column; occasionally dives are greater than 200 meters (Bolten, Riewald, and Bjorndal, unpublished data). In the vicinity of seamounts, oceanic banks or ridges that come close to the surface, or around oceanic islands, loggerheads may become epibenthic/demersal by feeding or spending time on the bottom. Turtles in Azorean waters can travel at sustained speeds of about 0.2 meters per second (Bolten, Riewald, and Bjorndal, unpublished data).

In Azorean waters, satellite telemetry data and flipper tag returns suggest a long period of residency (Bolten in press a), whereas turtles appear to be moving through Madeiran waters (Dellinger and Freitas 2000). This may not be surprising when one considers the physical oceanographic aspects of the regions. The Azorean region is characterized by a complexity of seamounts, banks, and the Mid-Atlantic Ridge, which results in a complexity of eddies and convergent zones – prime habitats for oceanic-stage loggerheads.

Diet

The diet of oceanic-stage loggerheads has been poorly studied. They are primarily carnivorous, although they do ingest some vegetation (Bjorndal 1997). Loggerheads in this life stage consume primarily coelenterates and salps, but also ingest a range of organisms including the pelagic snail *Janthina* spp., barnacles (*Lepas* spp.), and crabs (see Bjorndal 1997 for review).

Relationship of Oceanic Populations to Rookery Sources

Carr (1986) and later Bolten *et al*. (1993) used the comparison of size frequency distributions to suggest that the little loggerheads found in the oceanic zone around the Azores were an earlier life stage of the larger turtles in the neritic waters of the western Atlantic. The relationship between the little loggerheads in the oceanic zone and the larger-sized neritic loggerheads in the western Atlantic was further supported by a flipper tagging program managed by the Archie Carr Center for Sea Turtle Research at the University of Florida (Bolten in press a; Bolten *et al*. 1992a, b; Bjorndal *et al*. 1994). A number of turtles captured and tagged in the oceanic zone have been recaptured in the neritic zone of the western Atlantic (Bolten in press a).

With the development of molecular genetic tools (e.g., mitochondrial DNA sequence analyses), the relative contributions of rookeries to mixed stocks of oceanic-stage loggerheads could be evaluated (Bowen 1995, in press). After the Atlantic rookeries were genetically characterized by Encalada *et al*. (1998), Bolten *et al*. (1998) were able to demonstrate that the oceanic-stage loggerheads in the waters around the Azores and Madeira were primarily from rookeries in the southeastern U.S. (90 percent) and Mexico (10 percent).

Studies are currently underway with significantly larger sample sizes from the mixed oceanic-stage populations (Bolten *et al*., unpublished data); more complete rookery sampling (e.g., Cape Verde Islands, Luis Felipe *et al*., unpublished data); and increased sampling of southeastern U.S. rookeries (Pearce 2001; Bjorndal *et al*., unpublished data). These additional data will likely result in changes to the percentages of contributions from the specific rookeries but the conclusion that the primary source rookeries for the Azorean-Madeiran oceanic populations are from the western Atlantic (primarily the southeastern U.S.) will probably continue to be supported (Bolten *et al*., unpublished data). In addition, recent developments in statistical models for analyzing mixed stock composition will likely result in broader, and more realistic, confidence intervals for the point estimates of rookery contributions to foraging populations (Bolker *et al*. in press).

Based on flipper tag returns (Bolten in press, Bolten *et al*. 1992a) and on molecular genetic studies (Laurent *et al*. 1993, 1998), movement of little loggerheads from western Atlantic rookeries and Azorean waters into the western Mediterranean is probably more common than originally thought. These loggerheads from the western Atlantic apparently leave the Mediterranean before they mature and reproduce (Laurent *et al*. 1998).

Size Distribution

The size distribution of oceanic-stage loggerheads in the waters around the Azores ranges from 8.5 to 82 centimeters curved carapace length (CCL) with a mean of 34.5 +/- 12.6 centimeters (Bjorndal *et al*. 2000a). This size distribution is not significantly different from another nearby oceanic-zone aggregation in the waters around Madeira (Bolten *et al*. 1993).

Growth Rates

Length-frequency analyses generated the following estimates of the von Bertalanffy growth model: K = 0.072 +/- 0.003 yr^{-1} and asymptotic CCL (Linf) = 105.5 +/- 2.7 centimeters (Bjorndal *et al*. 2000a). The size-specific growth rate function from length-frequency analyses is consistent with growth rates calculated from recaptures of tagged turtles (summarized in Bjorndal *et al*. 2000a). Zug *et al*. (1995) evaluated the somatic growth rates of oceanic-stage loggerheads in the Pacific using skeletochronology. The age-specific growth function for the Pacific was similar in shape but with a slower growth rate than those for the Atlantic (Bjorndal *et al*. in review a).

Duration of the Oceanic Juvenile Stage

Using length-frequency analyses, Bjorndal *et al*. (2000a) estimated the duration of the oceanic stage to be 6.5 to 11.5 years depending on the size of the turtles when they leave the oceanic zone (46 to 64 centimeters CCL). Based on a skeletochronology study of neritic-stage loggerheads, Snover *et al*. (2000) concluded that loggerheads are 52 centimeters straight carapace length (SCL) when they settle in the neritic zone off the east coast of the United States. This value of 52 centimeters SCL is similar to the value of 53 centimeters CCL at the intersection of the cubic smoothing splines of the length frequency distributions of the oceanic stage and the neritic stage, which is equivalent to 8.2 years duration in the oceanic stage (Bjorndal *et al*. 2000a).

Survival Probabilities

Survival probabilities for the oceanic stage have been generated as fitted values in demographic models rather than direct estimates (Chaloupka in press, Heppell *et al*. in press). Catch-curve analyses can be used to estimate survival probabilities, but emigration and mortality are confounded. Bjorndal *et al*. (in review b) used catch-curve analyses to estimate survival probabilities of oceanic-stage loggerheads in the waters around the Azores. At ages before loggerheads begin to emigrate from the oceanic zone (2 to 6 years of age), the estimate of annual survival probability is 0.911. Turtles that are 2 to 6 years of age (18 to 44 centimeters CCL, Bjorndal *et al*. in review b) are not generally caught in the longline fishery (Bolten in press a).

After emigration begins at 7 years of age, the estimate of annual survival probability drops to 0.643, which is confounded by emigration but includes mortality from bycatch in longline fisheries (Bjorndal *et al*. in review b). Estimates of mortality prior to age 2 are not available, but mortality may be high during this stage both from natural predation and stochastic events that result in little loggerheads being passively swept into inappropriate habitats, such as Labrador or the waters around the British Isles (Carr 1986, Hays and Marsh 1997).

Juvenile Transitional Stage – Oceanic and Neritic Zones

The shift from the oceanic to the neritic zone is a dramatic one, and, as such, there is probably a period of transition, perhaps in both behavior and morphology. Kamezaki and Matsui (1997) discuss specific allometric relationships that change during the juvenile transitional stage that they suggest are related to changes in foraging behavior (e.g., epipelagic versus benthic). The geographic regions where the transitional stages occur may be in regions where major oceanic currents approach or enter the neritic zone. The broad size range over which the turtles in the Atlantic leave the oceanic zone and enter the neritic zone (Bjorndal *et al*. 2000a, 2001) may also suggest that this transitional stage is of variable duration. Factors that may drive this habitat shift (e.g., differential growth rates) are discussed in Bolten (in press a, b). Size frequency distributions of populations that fall between the oceanic stage and the neritic juvenile stage may support the existence of this transitional stage. The mean size of 53 centimeters CCL of a population off the Atlantic coast of Morocco is the estimated mid-point of the size distributions for the juvenile transitional stage and suggests that this population may represent a transitional stage between the oceanic and neritic stages (Tiwari *et al*. 2002). A juvenile transitional stage for the Mediterranean populations has also been suggested (Laurent *et al*. 1998). As Figure 8 indicates, if the oceanic-neritic transition is not complete, loggerheads may return to the oceanic zone. For example, a 78-centimeter loggerhead tagged along the east coast of Florida was recaptured in the Azores (Eckert and Martins 1989). Also, if juvenile loggerheads make multiple loops in the Atlantic Gyre system rather than a single developmental loop, this could result in periodic movements between the oceanic and neritic zones.

E.4. NERITIC ZONE: JUVENILE STAGE

In the western Atlantic, juvenile loggerheads recruit from oceanic pelagic to neritic demersal habitats when as small as 25 to 30 centimeters CCL, but most are 50 centimeters CCL at an age of about 7 to 10 years (Klinger and Musick 1995). In the western Atlantic, some demersal juvenile loggerheads make strong seasonal foraging migrations into temperate latitudes, occurring commonly as far north as Long Island, New York (Shoop and Kennedy 1992). Chesapeake Bay is a major seasonal developmental habitat in summer (Musick 1988) with 5,000 to 10,000 loggerheads present each summer (Byles 1988, Keinath *et al*. 1987). Around 95 percent of the loggerheads that visit Chesapeake Bay are juveniles. They usually enter the Bay in late May or early June when water temperatures rise to 16 to 18^{o}C, and depart from late September to early November (Lutcavage and Musick 1985, Keinath *et al*. 1987). Juveniles become resident for the summer along channel edges (5 to 13 meters) and forage back and forth along the bottom, passively with the tide within a home range of 10 to 80 square kilometers with preferred ranges (within the home range) of about 5 to 15 square kilometers (Byles 1988). These juveniles show strong foraging site fidelity. Animals displaced greater than 100 kilometers have returned to within a few kilometers of their point of origin within a few weeks (Byles 1988, Keinath *et al*. 1987). Of 121 loggerheads tagged in Virginia, 48 were recaptured there in subsequent (but not necessarily consecutive seasons) (Keinath 1993). Some were recaptured up to four seasons, showing strong foraging ground fidelity between seasons.

Recent research has shown that there are at least five subpopulations of loggerheads nesting in the western North Atlantic (Witzell *et al*. 2002). The South Florida Subpopulation, the largest subpopulation by far, nests along the Atlantic coast of Florida, followed by a Northern Subpopulation that nests from North Florida to Virginia (TEWG 2000). Norrgard and Graves (1995) found that of the juvenile loggerheads from the Chesapeake Bay, 64 percent were from the South Florida Subpopulation and 36 percent were from the Northern Subpopulation. Sears (1993, 1994) found a ratio of 50/50 from these two subpopulations of juvenile loggerheads from Charleston, South Carolina. These data suggest that both Virginia and the Georgia-South Carolina turtles selectively use more northerly summer developmental habitats than do Florida turtles. The South Florida Subpopulation is about nine times larger than the Northern Subpopulation, and the juvenile ratios should have been heavily skewed toward the South Florida Subpopulation if there were a random distribution of juveniles from both groups (Sears 1994). Aerial surveys and satellite tracking studies have shown that when juvenile loggerheads migrate from Chesapeake Bay in autumn, they travel relatively close to the coast (20 kilometers) and move south rounding Cape Hatteras around December (Keinath *et al*. 1996, Keinath 1993, Musick *et al*. 1994). They are joined in the fall by substantial numbers of juvenile loggerheads from the sounds of North Carolina (Epperly *et al*. 1995a, b). By December, most loggerheads have migrated south of Oregon Inlet, North Carolina (Musick *et al*. 1994), and by January most are south of Cape Hatteras (Epperly *et al*. 1995a). Most of the turtles that remain off North Carolina in January and February are found at the edge of the Gulf Stream (Epperly *et al*. 1995b). Keinath (1993) found that the juvenile loggerheads he tracked from Virginia wintered in two basically distinct areas: one group went down the coast, stayed inshore (hesitating during warm spells at various major inlets), and wintered inshore off southern Georgia and Florida, as far south as the Florida Keys. The other group followed the same inshore route south of Cape Hatteras to about Cape Lookout, North Carolina, and then moved offshore to winter on reefs along the shelf edge on the western side of the Gulf Stream.

Juvenile loggerheads of about the same size distribution (50 to 80 centimeters SCL) are common during the summer in estuaries as far south as Mosquito and Indian River Lagoons, Florida (Mendonca and Ehrhart 1982, Ehrhart 1983) and in the Gulf of Mexico (LeBuff 1990, Hildebrand 1982). In South Carolina and Georgia, they exhibit seasonal emigration from estuaries into the ocean in the autumn, with immigration occurring in the spring (Keinath *et al*. 1995). In the Mosquito Lagoon, Florida, juvenile loggerheads occur year-round, but are captured in smallest numbers during February and March (the time of lowest water temperature) (Mendonca and Ehrhart 1982). The reason for this may be because the turtles emigrate out into the ocean, or rather they may be inactive, actually bromating in the mud during this period. Juvenile loggerheads commonly brumate in winter by digging head first into the mud in the Canaveral Ship Channel (in the ocean not far from the Mosquito Lagoon) (Carr *et al*. 1980, Ogren and McVea 1982). Brumation has never been observed in Virginia (Musick and Limpus 1997) or in North Carolina (Epperly *et al*. 1995b), where winter water temperatures fall below the lethal lower limit for loggerheads (5 to 6.5^{o}C) (Schwartz 1978, Lutz *et al*.1989). Henwood

(1987) noted that juvenile loggerheads reached their peak abundance in winter in the Canaveral Ship Channel, and that some turtles tagged there in winter were recaptured to the north in summer.

E.5. NERITIC ZONE: ADULT STAGE

UNDER DEVELOPMENT

F. DEMOGRAPHY

UNDER DEVELOPMENT

G. THREATS

G.1. TERRESTRIAL ZONE (NESTING BEACH)

RESOURCE USE (NON-FISHERIES)

Illegal Harvest
In the United States, killing of nesting loggerheads is infrequent. However, on some beaches, human poaching of turtle nests and clandestine markets for eggs has been a problem (Ehrhart and Witherington 1987; Mark Dodd, Georgia Department of Natural Resources (GDNR), personal communication, 2000; Jorge Picon, FWS, personal communication, 2002). During the period 1983 to 1989, the Florida Marine Patrol made 29 arrests for illegal possession of turtle eggs (figure not apportioned by species). In Palm Beach, Martin, and St. Lucie Counties only (Florida coastal areas with what may be the highest prevalence of egg poaching), there were 33 arrests for possession or sale of sea turtle eggs from 1980 to 2002 (Captain Jeff Ardelean, FFWCC, personal communication, 2002).

Beach Cleaning
Beach raking to collect debris and trash may cause damage to nests. Several methods are used to clean beaches, including mechanical raking, hand raking, and picking up debris by hand. In mechanical raking, heavy machinery can repeatedly traverse nests and potentially compact the sand above them. Mann (1977) suggested that mortality within nests might increase when externally applied pressure from beach-cleaning machinery is common on soft beaches with large-grain sand. Beach cleaning vehicles also may leave ruts along the beach that hinder or trap emergent hatchlings (Hosier *et al*. 1981). Mechanically pulled rakes and hand rakes, particularly if the tongs are longer than 10 centimeters, penetrate the beach surface and may disturb incubating nests or uncover pre-emergent hatchlings near the surface of the nest.

Washback post-hatchlings (hatchlings that have left their nesting beaches, spent weeks or months at sea, and are then washed back onto the beach) are commonly stranded in seaweed washed in by late summer and fall storm events. Beach raking efforts to remove the seaweed may result in post-hatchlings being run over and crushed either as they remain tangled in the debris or as they attempt to return to the sea. In some areas, collected debris is buried directly on the beach, and this can lead to excavation and destruction of incubating egg clutches. Disposal of debris near the dune line or on the high beach can cover incubating egg clutches, hinder and entrap emergent hatchlings, and alter natural nest temperatures.

Human Presence

The greatest threat posed by humans on the beach at night is to disturb female turtles before they have completed their nesting. Up until the time a nesting sea turtle begins laying eggs, she may be easily frightened back into the ocean by human activity and lighting on the beach (McFarlane 1963). Once a turtle leaves the beach, she may return to the same location or select a new site later that night or the following night. However, repeated interruption of nesting may cause a turtle to construct her nest in a sub-optimum incubation environment, postpone nesting for several days, prompt movement many kilometers from the original chosen nesting site, and bring about the turtle shedding her eggs at sea (Murphy 1985). The extent to which heavy nighttime beach use by humans may affect sea turtle nesting range is not known.

Visitors using flashlights or lanterns or lighting campfires on the beach at night during the nesting season may deter nesting females from coming ashore and may disorient hatchlings (Mortimer 1989). Direct harassment may also cause adult turtles already on the beach to abandon their nesting activity or reduce the time spent covering the nest (Johnson *et al*. 1996).

It is unlawful for beach visitors to disturb sea turtle nests, hatchlings, or adults. Nevertheless, uninformed beachgoers, particularly children, have been reported digging into nests in search of eggs or hatchlings, presumably out of curiosity. Most often, though, impacts are indirect. For example, hatchlings may become trapped in holes dug on the beach. Additionally, research has shown that human footprints on the beach can interfere with the ability of hatchlings to reach the ocean (Hosier *et al*. 1981), and heavy pedestrian traffic may possibly compact sand over unmarked nests (Mann 1977), although the effect of this compaction has not been determined and may be negligible (Arianoutsou 1988). Depending on the nesting substrate, pedestrian traffic over nests near the time of emergence can cause the nests to collapse and result in hatchling mortality (Mann 1977, Dutton *et al*. 1994). Visitors are generally sympathetic to hatchlings and may pick them up and release them into the surf. The negative impacts of this activity may include some loss of imprinting to the beach.

Recreational Beach Equipment

The use and storage of lounge chairs, cabanas, umbrellas, catamarans, and other types of recreational equipment on nesting beaches can hamper or deter nesting by adult females and trap or impede hatchlings during their nest to sea migration. The documentation of false crawls at these obstacles is becoming increasingly common as more recreational beach equipment is left in place nightly on nesting beaches. Sobel (2002) describes nesting turtles being deterred by wooden lounge chairs that prevented access to the upper beach. Additionally, there are documented reports of nesting females being trapped under heavy wooden lounge chairs and cabanas, eggs being destroyed by equipment (e.g., beach umbrellas) penetrating the egg chamber, and hatchlings being hampered during emergence by equipment inadvertently placed on top of the nest (FFWCC, unpublished data).

Beach Vehicular Driving
The operation of public vehicles on nesting beaches for recreational purposes or beach access is allowed in northeast Florida (Nassau, Duval, St. Johns, Flagler, and Volusia Counties), northwest Florida (Walton and Gulf Counties), Georgia (all coastal islands, island residents only), and North Carolina (Emerald Isle, Cape Lookout National Seashore, Cape Hatteras National Seashore, and Currituck Banks). The operation of vehicles to conduct scientific research and management is allowed, with few exceptions, throughout the loggerhead's nesting range.

The presence of vehicles on the beach has the potential to negatively impact sea turtles by running over nesting females, hatchlings, stranded turtles that have washed ashore, and nests. In addition, the ruts left by vehicles in the sand may prevent or impede hatchlings from reaching the ocean following their emergence from the nest (Hosier *et al*.1981, Cox *et al*. 1994). Upon encountering a vehicle rut, hatchlings may be misoriented along the vehicle track, rather than crossing over it to reach the water. Apparently, hatchlings become diverted not because they cannot physically climb out of the rut (Hughes and Caine 1994), but because the sides of the track cast a shadow and the hatchlings lose their line of sight to the ocean horizon (Mann 1977). If hatchlings are detoured along the vehicle ruts, they are at greater risk of death from predation, fatigue, desiccation, and being crushed by additional vehicle traffic.

Vehicle lights and vehicle movement on the beach after dark can deter female sea turtles from nesting and disorient hatchlings. Sand compaction due to vehicles on the beach may hinder nest construction and hatchling emergence from nests. Driving directly above incubating egg clutches can cause sand compaction, which may decrease hatching success and directly kill pre-emergent hatchlings (Mann 1977). Additionally, vehicle traffic on nesting beaches may contribute to erosion, especially during high tides or on narrow beaches where driving is concentrated on the high beach and foredune.

Conservation and Research Activities
a. Nest Relocation
Relocation of sea turtle nests to higher beach sites or into hatcheries was once a recommended practice throughout the southeastern U.S. with the purpose of mitigating effects from erosion, vegetation encroachment, predation, and a variety of human-induced factors. The present emphasis of management is to be less manipulative with nests and hatchlings. This change in policy has followed an increased understanding of the potential adverse effects associated with nest relocation and restraint of hatchlings.

Although the philosophy of nest manipulation has changed recently, many nests are still relocated in the southeastern U.S. each year. On some Florida beaches, a high proportion of nests are moved from areas where artificial lighting threatens their hatchlings. For example, on densely developed Broward County beaches, approximately 1,700 loggerhead nests (66 percent of total nests) are excavated, transported, and relocated each year (1997-2001, FFWCC, unpublished data). Of these relocated nests, 13 percent are placed within hatcheries that restrain emerging hatchlings before they can be collected and released.

In Georgia through North Carolina, the principal reason for relocating nests has been to reduce the effects of erosion and inundation on nest survivorship, and the principal effect of relocation has been to redistribute nests to higher and drier areas of beach. Historically, nest manipulation was high in Georgia with managers relocating up to 70 percent of nests, but in recent years this proportion has been reduced (GDNR, unpublished data). In 1999, 2000, and 2001, respectively, 31.6, 28.7, and 27.7 percent of loggerhead nests were relocated in Georgia. All hatcheries in Georgia were eliminated in 2000, and relocated nests are now moved to the closest suitable location in the dunes. In South Carolina, there has been a similar trend. In 1997, 1998, 1999, 2000, and 2001, respectively, 58, 56, 44, 38, and 44 percent of loggerhead nests were relocated away from the surf, some into beach hatcheries (South Carolina Department of Natural Resources (SCDNR), unpublished data). In North Carolina between 1997 and 2001, approximately 45 percent of all nests were relocated (NCWRC, unpublished data).

Although some relocated nests produce more hatchlings than they would have at the location where the turtles put them, all relocated nests risk a number of effects from manipulation. Moody (1998) found that at 10 of 12 Florida beaches surveyed in 1993 and 1994, nests relocated for conservation purposes had significantly lower hatchling emergence rates than nests left *in situ*. Movement alone is known to kill developing embryos by disrupting delicate membranes that attach to the inside of the egg (Limpus *et al*. 1979, Parmenter 1980). Individual egg loss in association with handling and transport of relocated nests has also been reported. Such an incident can result in fluids contaminating the nest and invite fungal and bacterial growth. Nest relocation can have adverse impacts on incubation temperature and hence sex ratios (Spotila *et al*. 1983), and on gas exchange, hydric environment, hatching success, and hatchling emergence. Relocating nests into sands deficient in oxygen or moisture can result in mortality, morbidity, and reduced behavioral competence of hatchlings (Ackerman 1980, Packard *et al*. 1981, Packard *et al*. 1984, Packard *et al*. 1985, Packard and Packard 1986, Miller *et al*. 1987, Packard *et al*. 1988, McGehee 1990). Furthermore, relocating nests may concentrate eggs in an area resulting in a greater susceptibility to catastrophic events and predation from both land and marine predators (Glenn 1998, Wyneken *et al*. 1998). Delaying hatchlings from crawling down the beach significantly lowers their crawling and swimming speeds (Hewavisenthi 1994, Pilcher and Enderby 2001) and may increase the risk of predation.

Although sea turtle management guidelines of the southeastern states specify nest relocation as a practice of last resort, it is likely that a high proportion of nests are unnecessarily relocated. That is, the nests would have been productive at their original location. Occasional tidal inundation of nests appears to have minimal effect on loggerhead reproductive success (McGehee 1990, Foley 1998). Nests located where there are threats from beachfront lighting, foot traffic, and mammalian predation, can be protected by addressing the threat directly or by protecting the nest site rather than by moving the nest.

b. Nesting Surveys

Although nighttime nesting surveys have been implicated in reducing nesting on Kiawah Island, South Carolina (Murphy *et al*. 1999), most research and monitoring activity is likely to have a minimal effect on nesting turtles and hatchlings. In Florida, surveyors who use vehicles for nesting surveys monitor the beach during early morning when encounters with turtles are unlikely. These surveyors also use all-terrain motorcycles with low-pressure tires that do not leave ruts and that exert less pressure than a human footprint. Florida surveyors are also advised during annual training workshops on how to avoid harmful encounters with sea turtles, their nests, shorebirds, and other sensitive beach features.

CONSTRUCTION AND DEVELOPMENT

Beach Sand Placement

Beach sand placement refers to beach restoration, beach nourishment, and inlet sand bypassing projects that are carried out to provide a temporary remedy for erosion along the coast. Beach restoration is the placement of sand along the shoreline to rebuild a beach that has been totally lost to erosion. Beach nourishment is the periodic replenishment of a restored beach to maintain a desired beach width for protection of coastal structures. Beach nourishment often involves the excavation of large quantities of sand from one site and placing it on an existing, but eroding, section of coastline. Sand is most typically dredged from inlets or offshore "borrow" areas, although inland sand sources may also be used. Inlet sand bypassing involves removing sand from an inlet for navigational purposes and often involves disposal of the material onto a beach. Inlet sand bypass systems are engineered to allow sand that has been restricted from its normal movement pattern by a man-made structure (jetty or artificially deepened channel) to be placed on the downdrift beach. These systems usually consist of a large depression constructed near the end of a jetty or groin on the updrift side of an inlet. As sand migrates past the structure, it collects in the sink. When the sink is full, sand is pumped to the downdrift beach with a hydraulic dredge.

Beach sand placement is generally viewed as less harmful to sea turtles than armoring, but it too can affect sea turtle reproductive success in a variety of ways. Although the placement of sand on beaches may provide a greater quantity of nesting habitat, the quality of that habitat may be less suitable than pre-existing natural beaches. Sub-optimal nesting habitat may cause decreased nesting success, place an increased energy burden on nesting females, result in abnormal nest construction, and reduce the survivorship of eggs and hatchlings. Crain *et al*. (1995) provides a review of the potential effects of beach nourishment on sea turtles.

Construction related impacts of sand placement projects that are performed during the nesting and hatching season can occur. Pipelines and heavy equipment can create barriers to nesting females, causing a higher incidence of false crawls (non-nesting emergences). Increased human activity on the project beach at night may cause further disturbance to nesting females. Unmarked nests may be crushed by construction equipment or buried during sand placement. Nests relocated to a beach site outside the project area may experience reduced reproductive success (Limpus *et al*. 1979, Moody 1998). Project lighting along the beach and in the nearshore area of a borrow site may deter nesting females and misorient emergent hatchlings from adjacent non-project beaches.

Constructed beaches tend to differ from natural beaches in several important ways. They are typically wider, flatter, more compact, and the sediments are moister than those on natural beaches (Nelson *et al*. 1987, Ackerman *et al*. 1991, Ernest and Martin 1999). On severely eroded sections of beach, where little or no suitable nesting habitat previously existed, sand placement can result in increased nesting (Ernest and Martin 1999). However, on most beaches, nesting success typically declines for the first year or two following construction, even though more nesting habitat is available for turtles (Trindell *et al*. 1998, Ernest and Martin 1999, Herren 1999). Reduced nesting success on constructed beaches has been attributed to increased sand compaction, escarpment formation, and changes in beach profile (Nelson *et al*. 1987, Crain *et al*. 1995, Lutcavage *et al*. 1997, Steinitz *et al*. 1998, Ernest and Martin 1999, Rumbold *et al*. 2001). Compaction can inhibit nest construction or increase the amount of time it takes for turtles to construct nests, while escarpments often cause female turtles to return to the ocean without nesting or to deposit their nests seaward of the escarpment where they are more susceptible to frequent and prolonged tidal inundation.

Beach sand placement can affect the incubation environment of nests by altering the moisture content, gas exchange, and temperature of sediments (Ackerman *et al*. 1991, Ackerman 1997, Parkinson and Magron 1998). The extent to which the incubation environment is altered is largely dependent on the similarity of the placed sands and the natural sediments they replace. Consequently, the results of studies assessing the effects of sand placement on reproductive success have varied among study sites.

Even though constructed beaches are wider, nests deposited there may experience higher rates of wash out than those on relatively narrow, steeply sloped beaches (Ernest and Martin 1999). This occurs because nests on constructed beaches are more broadly distributed than those on natural beaches, where they tend to be clustered near the base of the dune. Nests laid closest to the waterline on constructed beaches may be lost during the first year or two following construction as the beach undergoes an equilibration process during which seaward portions of the beach are lost to erosion.

Placement of sand on highly eroded beaches, especially those with a complete absence of dry beach, can be beneficial to nesting turtles if conducted properly. Consideration of sea turtle concerns in project planning must be made to ensure the sand source is compatible with naturally occurring beach sediments in the area (in terms of grain size, shape, color, etc.) and that remediation measures are incorporated into the project to allow for successful nesting, nest incubation, and hatchling emergence. Beach and dune profiles that mimic the beaches nesting loggerheads prefer (narrow and steeply sloped with a prominent vegetated dune, Provancha and Ehrhart 1987) are seldom the choice for sand placement projects. Rather, constructed beaches are commonly engineered to be wide and flat, traits that achieve the principal goals of upland property protection and increased area for human visitation.

Although sand bypassing efforts have the potential to reduce downdrift erosion effects, there may be effects on sea turtle reproduction that are similar to those from beach nourishment. For example, several researchers have evaluated the effects of a sand bypassing program on sea turtle reproductive success at Sebastian Inlet on Florida's Atlantic Coast. The first of those studies detected no significant differences in hatchling emergence success between the beach receiving bypassed sand and a control beach farther downdrift (Ryder 1993). However, in a study of a subsequent sand bypass effort, Herren (1999) found a significant reduction in hatchling emerging success on the nourished beaches compared to a control. Differences in results between studies probably relate to variability in the characteristics of sediments placed on the beach. In addition to reduced reproductive success, Herren (1999) also noted a decline in nesting success downdrift of the inlet during the first year or two following a sand bypass project, likely caused by the presence of escarpments that formed on the beach post-construction. Witherington *et al*. (in press) measured loggerhead nesting density over a 12-year period (1989 to 2000) within 6 kilometers of Sebastian Inlet and found nesting to decrease significantly with proximity to the inlet in both updrift and downdrift directions.

Beach Armoring

Armoring is any rigid structure placed parallel to the shoreline on the upper beach to prevent both landward retreat of the shoreline and inundation or loss of upland property by flooding and wave action (Kraus and McDougal 1996). Armoring includes bulkheads, seawalls, soil retaining walls, rock revetments, sandbags, and geotextile tubes. Schroeder and Mosier (2000) provide descriptions of these different structures. Although armoring structures are generally effective in protecting beachfront property, they do little to promote or maintain sandy beaches. These structures have the potential to influence natural shoreline processes and the physical beach environment, but the effects are not well understood. It is clear that armoring structures prevent long-term recovery of the beach/dune system (i.e., building of the back beach) by physically prohibiting dune formation from wave uprush and wind-blown sand. However, reported topographic effects seaward and adjacent to these structures vary between project sites (Kaufman and Pilkey 1979, Pilkey *et al*. 1984, Kraus 1988, Kraus and McDougal 1996).

Erosion of adjacent downdrift beaches can occur if an updrift armoring structure acts as a jetty and impounds sand (Kraus 1988, Tait and Griggs 1990). Additionally, these structures can cause wave reflection and scour, processes that accelerate erosion seaward of the structure and that steepen the offshore profile (Pilkey *et al*. 1984). Sand can move alongshore past an armoring structure, but it is not clear whether the longshore sediment transport rate changes (Kraus and McDougal 1996). Pilkey *et al*. (1984) contend that the intensity of longshore currents does increase in front of armoring structures and this hastens removal of beach sand. Most likely, the extent to which any of these potentially harmful effects may be realized is largely dependent upon a structure's physical position on the beach relative to the surf zone (Kraus 1988, Tait and Griggs 1990). The closer an armoring structure is to the surf zone, the greater its potential for altering shoreline processes.

Considerable anecdotal information suggests that permanent armoring structures can diminish the quality of sea turtle nesting habitat. However, there have been few experimental studies designed specifically to assess the impacts of these structures on sea turtle nesting. Mosier (1998) and Mosier and Witherington (2002) recorded the behavior of nesting turtles in front of seawalls and adjacent unarmored sections of beach. Mosier (1998) reported that fewer loggerheads made nesting attempts on beaches fronted by seawalls than on adjacent beaches where armoring structures were absent. Both studies found that when turtles did emerge in the presence of armoring structures, more returned to the water without nesting than those on non-armored beaches. Additionally, Mosier (1998) found that turtles on armored sections of beach tended to wander greater distances than those that emerged on adjacent natural beaches. It is unknown if this additional energy expenditure reduces reproductive output. Armoring structures can effectively eliminate a turtle's access to upper regions of the beach/dune system. Consequently, nests on armored beaches were generally found at lower elevations than those on non-walled beaches. Lower elevations subject nests to a greater risk of repeated tidal inundation and erosion and can potentially alter thermal regimes, an important factor in determining the sex ratio of hatchlings (Mrosovsky and Provancha 1989, Mrosovsky 1994, Ackerman 1997, Delpech and Foote 1998). The negative effects of armoring become more pronounced the closer the structures are to the surf zone. Thus, the quality of beach habitat seaward of armoring structures on eroding sections of coastline can be expected to diminish as the shoreline recedes.

Impacts also can occur if the installation of structures takes place during the sea turtle nesting season. Unmarked nests can be crushed or uncovered by heavy equipment. Vibrations and water runoff from jetting operations during installation of structures can damage nests as well. There have also been reported incidents of nesting turtles and hatchlings getting caught in construction debris or trapped in excavations at the construction site (FFWCC, unpublished data). In addition, hatchlings have been trapped in holes or crevices of exposed riprap and geotextile tubes. Both nesting turtles and hatchlings have been entangled or entrapped in the debris of failed structures. There have also been reports of injuries and deaths of nesting turtles that have fallen from seawalls after crawling onto them from adjacent properties (FFWCC, unpublished data).

As the extent of armoring on beaches increases, the probability of a nesting turtle encountering a seawall or depositing a nest in sub-optimal habitat increases. The proportion of coastline that is armored is approximately 18 percent (239 kilometers) in Florida (Clark 1992; Schroeder and Mosier 2000), 9 percent (14 kilometers) in Georgia (Mark Dodd, GDNR, personal communication, 2000), 12 percent (29 kilometers) in South Carolina (Sally Murphy, SCDNR, personal communication, 2000), and 2 percent (9 kilometers) in North Carolina (Sean McGuire, North Carolina Division of Coastal Management, personal communication, 2002). These assessments of armoring extent do not include structures that are a barrier to sea turtle nesting but that do not fit the definition of armoring, such as dune crossovers, cabanas, sand fences, and recreational equipment.

Other Shoreline Stabilizations

a. Groins and Jetties

Groins and jetties are shore-perpendicular structures that are designed to trap sand that would otherwise be transported by longshore currents. Jetties are specifically placed to keep sand from flowing into channels (Kaufman and Pilkey 1979, Komar 1983). In preventing normal sand transport, these structures accrete updrift beaches while causing accelerated beach erosion downdrift of the structures (Komar 1983, Pilkey *et al*. 1984, National Research Council 1987), a process that results in degradation of sea turtle nesting habitat. As sand fills the area updrift from the groin or jetty, some littoral drift and sand deposition on adjacent downdrift beaches may occur due to spillover. However, these structures often force the stream of sand into deeper offshore water where it is lost from the system (Kaufman and Pilkey 1979). The greatest changes in beach profile near groins and jetties are observed close to the structures, but effects eventually may extend many kilometers along the coast (Komar 1983). Beach nourishment only temporarily alleviates effects of groin construction on downdrift beaches (Komar 1983).

Jetties are placed at ocean inlets to keep transported sand from closing the inlet channel. Together, jetties and inlets are known to have profound effects on adjacent beaches (Kaufman and Pilkey 1979). Witherington *et al*. (in press) found a significant negative relationship between loggerhead nesting density and distance from the nearest of 17 ocean inlets on the Atlantic coast of Florida. The effect of inlets in lowering nesting density was observed both updrift and downdrift of the inlets, leading the researchers to propose that beach instability from both erosion and accretion may discourage loggerhead nesting.

Construction of these structures during the nesting season may result in the destruction of nests, disturbance of females attempting to nest, and disorientation of emerging hatchlings from project lighting. Following construction, the presence of groins and jetties may interfere with nesting turtle access to the beach, result in a change in beach profile and width (downdrift erosion, loss of sandy berms, and escarpment formation), trap hatchlings, and concentrate predatory fishes, resulting in higher probabilities of hatchling predation as hatchlings enter the ocean and begin their offshore migration. Testing of an experimental mesh groin system is now underway to determine how successful it may be as a beach restoration system. The mesh groin system is a temporary set of groins composed of net "fences" that have the potential to entrap hatchlings within the openings of the net or within folds created by the billowing or bagging of the net material.

Escarpments may develop on beaches between groins as the beaches equilibrate to their final profiles. These escarpments are known to prevent female turtles from nesting on the upper beach and can cause turtles to choose unsuitable nesting areas to deposit eggs, such as seaward of an escarpment. These nest sites commonly receive prolonged tidal inundation and erosion that results in nest failure (Nelson and Blihovde 1998).

As the groin structures fail and break apart, they spread debris on the beach, which may further impede nesting females from accessing suitable nesting sites and trap both hatchlings and nesting turtles. Geotextile tubes begin to disintegrate when exposed to ultraviolet light (life expectancy is approximately 5 to 10 years). This may result in pieces of geotextile material, a woven plastic-like substance, floating off or being washed up on the beach. Although painting the exposed portions of the geotextile tube to protect them from ultraviolet light will slow down the rate of disintegration, it will still occur. The material may be ingested by sea turtles or entangle them, either of which could result in death.

b. Offshore Breakwaters

Breakwaters are typically constructed from rock or concrete units and are placed in nearshore waters to protect the shoreline by reducing wave energy (National Research Council 1990b). This reduction in wave energy modifies the longshore transport of sand and may result in an accumulation of sand and a reduction in erosion along the shoreline adjacent to the breakwater. However, the placement of breakwaters may result in the formation of a sand projection that connects the beach to the breakwater as sand accumulates. This creates a situation where the breakwater acts as a headland rather than an offshore feature. The breakwater then functions as a barrier to the longshore transport of material in a manner similar to a groin, resulting in downdrift erosion (National Research Council 1995) and degradation of downdrift sea turtle nesting habitat.

Breakwaters may be built with different top elevations. They may be built to project above the water's surface, or they may be built as submerged structures that are designed to reduce the height of waves but not to absorb or reflect all wave energy (National Research Council 1989). Emergent breakwaters that are oriented parallel to the shoreline have the potential to interfere with the movement of adult females to and from the nesting beach; function as barriers to sea turtle hatchlings during their offshore migration; entrap hatchlings in the crevices of the structures or within eddies or other currents associated with the structures; and increase hatchling and adult female energy expenditure in their attempts to bypass the structures.

The presence of breakwaters also has the potential to attract and concentrate predatory fishes and to provide perching spots for predatory birds, resulting in higher probabilities of hatchling predation as hatchlings begin their offshore migration. Hatchling predation in nearshore waters is variable and is observed to be considerably higher near submerged structures such as reefs (Witherington and Salmon 1992, Gyuris 1994, Wyneken and Salmon 1996). There are many documented occurrences of nearshore predators captured with hatchlings found in their digestive tracts (Stancyk 1995). During hatchling predation studies in Broward County, Florida, it was documented that predatory fish species targeted sea turtle hatchlings and learned where to concentrate foraging efforts (Glenn 1998, Wyneken *et al*. 1998).

Sand Fences

Sand fences, also known as snow fences and drift fences, are erected to build and stabilize dunes by trapping sand moving along the beach and by preventing excessive sand loss. Additionally, these fences can serve to protect dune systems by deterring foot traffic. Sand fences are constructed of narrowly spaced wooden or plastic slats or plastic fabric, and if improperly placed can trap hatchling turtles and act as barriers to both nesting and hatchling turtles (National Research Council 1990a). In Florida, Georgia, South Carolina, and North Carolina, the design and placement of sand fences on the beach are State-regulated to avoid negative impacts to sea turtles. One requirement is to position sand fences in multiple, short, shore-oblique lines rather than a single, long, shore-parallel line.

Stormwater Outfalls

Rainfall on the dunes and beaches percolates rapidly into the permeable sands and produces little, if any, runoff. However, runoff from beachfront parking lots, building rooftops, roads, decks, and draining swimming pools adjacent to the beach may be discharged directly to the beaches and dunes either by sheet flow, through stormwater collection system outfalls, or through small diameter pipes. These outfalls are known to create localized erosion channels, prevent natural dune establishment, and wash out sea turtle nests (FFWCC, unpublished data).

Coastal Construction

In addition to shoreline protection activities, there are a variety of other coastal construction activities that may affect sea turtles. These include construction, repair, and maintenance of upland structures and dune crossovers, installation of utility cables, installation and repair of public infrastructure (such as coastal highways and emergency evacuation routes), dune restoration, and vehicular traffic and lighting associated with any of these activities. Many of these activities may alter nesting habitat and harm sea turtle nests, adults, and hatchlings as described previously for coastal armoring. Most direct construction-related impacts can be avoided by requiring that non-emergency activities be performed outside of the nesting and hatching season.

ECOSYSTEM ALTERATIONS

Beach Erosion and Accretion
Erosion, frequent or prolonged tidal inundation, and accretion can negatively affect incubating egg clutches. Short-term erosion events (e.g., atmospheric fronts, northeasters, tropical storms, and hurricanes) are common phenomena throughout the loggerhead nesting range and may vary considerably from year to year. Sea turtles have evolved a strategy to offset these natural events by laying large numbers of eggs and by distributing their nests both spatially and temporally. Thus, rarely is the total annual hatchling production affected by storm-generated beach erosion and inundation. However, human activities along coastlines can accelerate erosion rates, interrupt natural shoreline migration, and reduce both the quantity and quality of available nesting habitat. It is unclear to what extent these human-induced effects might lower hatchling productivity.

During erosion events, nests may be uncovered or completely washed away. Nests that are not washed away may suffer reduced reproductive success as the result of frequent or prolonged tidal inundation. Eggs saturated with seawater are susceptible to embryonic mortality (Bustard and Greenham 1968, Milton *et al*. 1994, Martin 1996). However, in spite of the potential for reduced hatching success, Foley (1998) found that loggerhead eggs can successfully survive periodic tidal inundation. For instance, one nest with an emergence success of 70 percent was completely inundated at least twice, inundated at the middle-clutch level at least four other times, and inundated at the lower-clutch level at least four other times. Another nest with an emergence success of 92.8 percent was inundated at the middle-clutch level at least twice and at the lower-clutch level at least one other time. Similarly, Ernest and Martin (1993) found that although frequent or prolonged tidal inundation resulted in fewer emergent hatchlings, occasional overwash of nests appeared to have minimal effect on reproductive success.

Accretion of sand above incubating nests may also result in egg and hatchling mortality. Ehrhart and Witherington (1987) found that accretion of sand over loggerhead clutches killed all embryos in affected nests, presumably from suffocation.

POLLUTION

Oil Pollution
Oil spills in the vicinity of nesting beaches just prior to or during the sea turtle nesting season could place nesting females, incubating egg clutches, and hatchlings at significant risk (Fritts and McGehee 1982, Lutcavage *et al*. 1997, Witherington 1999). Fritts and McGehee (1982) conducted both field and laboratory studies to determine the effects of petroleum on the development and survival of sea turtle embryos. Their results suggest that an oil spill resulting in contamination of nesting beaches before the nesting season may affect nesting success for only a short period, if at all, but a spill resulting in the deposition of oil on eggs or on top of an incubating nest is likely to increase mortality and result in abnormal development of hatchlings. They concluded that the overall effect of oil spills on turtles was likely to be dependent on the timing of the spill and the age of the oil.

Two recent oil spills that occurred near loggerhead nesting beaches in Florida were observed to affect eggs, hatchlings, and nesting females. A spill involving approximately 350,000 gallons of fuel oil spilled in Tampa Bay in August 1993 and was carried onto nesting beaches on Pinellas County. Approximately 212 hatchlings were killed and 2,177 eggs and hatchlings were injured (Florida Department of Environmental Protection *et al*. 1997). Another spill near the beaches of Broward County in August 2000 involved approximately 15,000 gallons of oil and tar (National Oceanic and Atmospheric Administration and Florida Department of Environmental Protection 2002). Estimates from models were that approximately 1,500 to 2,000 hatchlings and 0 to 1 adults were injured or killed.

Oil cleanup activities can also be harmful. Earth-moving equipment can dissuade females from nesting and destroy nests, containment booms can entrap hatchlings, and lighting from nighttime activities can misdirect turtles (Witherington 1999).

Coastal Lighting

Both nesting and hatchling sea turtles are adversely affected by the presence of artificial lighting on or near the beach (Witherington and Martin 2000). Experimental studies have shown that artificial lighting deters adult female turtles from emerging from the ocean to nest (Witherington 1992). Witherington (1986) noted that loggerheads aborted nesting attempts at a greater frequency in lighted areas. Because adult females rely on visual brightness cues to find their way back to the ocean after nesting, those turtles that nest on lighted beaches may be disoriented by artificial lighting and have difficulty finding their way back to the ocean. In the lighted-beach experiments described by Witherington (1992), only a few turtles returning to the sea after nesting were misdirected by lighting, but those misdirected turtles were observed to have spent a large portion of the night wandering in search of the ocean. In some cases, misdirected nesting females have crawled onto coastal highways and have been struck and killed by vehicles (FFWCC, unpublished data).

Hatchling sea turtles exhibit a robust sea-finding behavior guided by visual cues (Witherington and Bjorndal 1991, Salmon *et al*. 1992, Lohmann *et al*. 1997, Witherington and Martin 2000, Lohmann and Lohmann in press), and direct and timely migration from the nest to sea appears critical to their survivorship. Although the mechanism involved in sea finding is complex, involving cues from both brightness and shape, it is clear that strong brightness stimuli can override other competing cues (Witherington and Martin 2000).

Hatchlings have a tendency to orient toward the brightest direction as integrated over a broad horizontal area. On natural undeveloped beaches, the brightest direction is commonly away from elevated shapes (e.g., dune, vegetation, etc.) and their silhouettes and toward the broad open horizon of the sea. On developed beaches, the brightest direction is often away from the ocean and toward lighted structures. Hatchlings unable to find the ocean, or delayed in reaching it, are likely to incur high mortality from dehydration, exhaustion, or predation (Carr and Ogren 1960, Ehrhart and Witherington 1987, Witherington and Martin 2000). Hatchlings lured into lighted parking lots or toward streetlights are often crushed by passing vehicles (McFarlane 1963, Philibosian 1976, Peters and Verhoeven 1994, Witherington and Martin 2000). Uncommonly intense artificial lighting can draw hatchlings back out of the surf (Daniel and Smith 1947, Carr and Ogren 1960, Ehrhart and Witherington 1987).

Although the attributes that can make a light source harmful to sea turtles are complex, a simple rule has proven useful in identifying lights that pose potential problems for sea turtles. Witherington and Martin (2000) propose that artificial light sources are “likely to cause problems for sea turtles if light from the source can be seen by an observer standing anywhere on the beach.” This visible light can come directly from any glowing portion of a luminaire, including the lamp, globe, or reflector, or indirectly by reflection from buildings or trees that are visible from the beach. Bright or numerous light sources, especially those directed upward, will illuminate sea mist and low clouds, creating a distinct sky glow visible from the beach. Field research suggests hatchling orientation can be disrupted by the sky glow from heavily lighted coastal areas even when no direct lighting is visible (Witherington *et al*. 1994).

The ephemeral nature of evidence from hatchling disorientation and mortality makes it difficult to accurately assess how many hatchlings are misdirected and killed by artificial lighting. Reports of hatchling disorientation events in Florida describe several hundred nests each year and are likely to involve tens of thousands of hatchlings (Nelson *et al*. 2002). However, this number calculated from disorientation reports is likely to be a vast underestimate. Independent of these reports, Witherington *et al*. (1996) surveyed hatchling orientation at nests located at 23 representative beaches in six counties around Florida in 1993 and1994 and found that, by county, approximately 10 to 30 percent of nests showed evidence of hatchlings disoriented by lighting. From this survey and from measures of hatchling production (FFWCC, unpublished data), the number of hatchlings disoriented by lighting in Florida is calculated to be in the range of hundreds of thousands per year.

Beach Debris

Hatchlings often must navigate through a variety of obstacles before reaching the ocean. These include natural and human-made debris. Debris on the beach may interfere with a hatchling’s progress toward the ocean. Research has shown that travel times of hatchlings from the nest to the water may be extended when traversing areas of heavy foot traffic or vehicular ruts (Hosier et al. 1981); the same is true of debris on beach. Hatchlings may be upended and spend both time and energy in righting themselves. Some beach debris may have the potential to trap hatchlings and prevent them from successfully reaching the ocean. In addition, debris over the top of nests may impede or prevent hatchling emergence.

SPECIES INTERACTIONS

Predation

Depredation of sea turtle eggs and hatchlings by natural and introduced species occurs on almost all nesting beaches. The most common predators in the southeastern United States are ghost crabs (*Ocypode quadrata*), raccoons (*Procyon lotor*), feral hogs (*Sus scrofa*), foxes (*Urocyon cinereoargenteus* and *Vulpes vulpes*), coyotes (*Canis latrans*), armadillos (*Dasypus novemcinctus*), and fire ants (*Solenopsis* spp.) (Dodd 1988, Stancyk 1995). Raccoons are particularly destructive and may take up to 96 percent of all nests deposited on a beach (Davis and Whiting 1977, Hopkins and Murphy 1980, Stancyk *et al*. 1980, Talbert *et al*. 1980, Schroeder 1981, Labisky *et al*. 1986). Prior to hog control efforts, up to 45 percent of all nests deposited at the Cape Canaveral Air Force Station, Florida, were depredated by feral hogs

(FFWCC, unpublished data). In 1990, an estimated 70 percent of loggerhead nests were destroyed by feral hogs on Ossabaw Island, Georgia, prior to the implementation of predator control programs (GDNR, unpublished data). In addition to the destruction of eggs, certain predators may take considerable numbers of hatchlings just prior to or upon emergence from the sand.

Although not considered a typical form of predation, roots of sea oats, railroad vine, and other dune plants sometimes invade the nest cavity and penetrate incubating eggs (Witherington 1986). This occurs primarily in nests laid high on the beach at or landward of the toe of the dune.

Exotic Dune and Beach Vegetation
Non-native vegetation has invaded many coastal areas and often outcompetes native species such as sea oats (*Uniola paniculata*), railroad vine (*Ipomoea pes-caprae*), sea grape (*Coccoloba uvifera*), bitter panicgrass (*Panicum amarum*), and seaside pennywort (*Hydrocotyle bonariensis*). The invasion of less stabilizing vegetation can lead to increased erosion and degradation of suitable nesting habitat. Exotic vegetation may also form impenetrable root mats that can prevent proper nest cavity excavation, invade and desiccate eggs, or trap hatchlings.

The Australian pine (*Casuarina equisetifolia*) is particularly harmful to sea turtles. Dense stands have taken over many coastal areas throughout central and south Florida. Australian pines cause excessive shading of the beach that would not otherwise occur. Studies in Florida suggest that nests laid in shaded areas are subjected to lower incubation temperatures, which may alter the natural hatchling sex ratio (Marcus and Maley 1987, Schmelz and Mezich 1988, Hanson *et al*. 1998). Fallen Australian pines limit access to suitable nest sites and can entrap nesting females (Austin 1978, Reardon and Mansfield 1997). The shallow root network of these pines can interfere with nest construction (Schmelz and Mezich 1988). Davis and Whiting (1977) reported that nesting activity declined in Everglades National Park where dense stands of Australian pine took over native dune vegetation on a remote nesting beach.

OTHER FACTORS

Natural Catastrophes
Short-term erosion events (e.g., atmospheric fronts, northeasters, tropical storms, and hurricanes) are common phenomena throughout the loggerhead nesting range and may vary considerably from year to year. Sea turtles have evolved a strategy to offset these natural events by laying large numbers of eggs and by distributing their nests both spatially and temporally. Thus, rarely is the total annual hatchling production affected by storm-generated beach erosion and inundation.

G.2. NERITIC ZONE

RESOURCE USE (FISHERIES)

Trawl Fisheries

Of all commercial and recreational fisheries in the United States, shrimp trawling is the most damaging to the recovery of sea turtle populations. In a 1990 study, the National Academy of Sciences estimated that between 5,000 and 50,000 loggerheads were killed annually by the offshore shrimping fleet in the southeastern United States Atlantic and Gulf of Mexico (National Research Council 1990). Mortality associated with shrimp trawls was estimated to be 10 times that of all other human-related factors combined. Most of these turtles were neritic juveniles and subadults, the life stages most critical to the stability and recovery of sea turtle populations (Crouse *et al*. 1987, Crowder *et al*. 1994).

In 1978, NMFS initiated the development of a net modification to allow captured turtles to escape from shrimp nets. The original modification, known as a Turtle Excluder Device (TED), was a large cage-like design that was effective in excluding turtles, but proved to be unwieldy and potentially dangerous for fishermen. NMFS, in cooperation with commercial fishermen, developed several new lighter TED designs based on net modifications commonly used by shrimpers to reduce unwanted bycatch (e.g., cannonball jellyfish). These designs consisted of a metal-grid or webbing ramp that directed turtles to an escape opening cut in the top or bottom of the net. Because of the increasing number of new TED designs developed by fishermen, NMFS adopted standardized guidelines that required all approved TEDs to be 97 percent effective in excluding turtles (NMFS 1987; 52 FR 24244).

By 1986, lack of voluntary widespread use of TEDs by shrimpers resulted in Federal regulations requiring their implementation (Epperly 2002). NMFS published final regulations in June 1987; however, implementation was delayed as a result of legal and congressional action. This delay prompted several states, including South Carolina and Florida, to require the use of TEDs in waters under their jurisdiction. The use of TEDs for all offshore trawlers from North Carolina to Texas was implemented in September 1989. By December 1994, regulations requiring fishery-wide use of TEDs, including all inshore areas, were in place (Epperly 2002).

The overall effectiveness of TEDs is difficult to assess. Crowder *et al*. (1995) found sea turtle strandings were reduced by 44 percent in South Carolina following the implementation of TEDs. Royle and Crowder (1998) found strandings were reduced by 40 percent in South Carolina and 58 percent in Georgia over the period from 1980-1997. By contrast, Shoop *et al*. (1999) found no reduction in sea turtle strandings on Cumberland Island, Georgia, with increased TED use. In the Gulf of Mexico, Caillouet *et al*. (1996) found that TED regulations did not diminish the statistical correlation between stranding rates and commercial fishing intensity. Large-scale stranding events associated with shrimp fishing activity have been documented following the mandated use of TEDs.

Although TEDs were found to reduce trawl related sea turtle mortality in some cases, the use of inefficient designs and small opening sizes has reduced their potential effectiveness. For example, several TED configurations were approved for use by NMFS (e.g., Morrison soft TED), but were later disallowed when additional testing found they captured and drowned turtles. Epperly and Teas (2002) found minimum TED opening sizes were too small to allow large subadult and adult loggerheads to escape from shrimp trawls. They estimated that 33 to 47 percent of the total loggerhead strandings measured on U.S. beaches from 1986 to1999 were too large to fit through the minimum TED opening.

Several other shrimp-fishery related factors have been implicated in contributing to high loggerhead mortality rates. These factors include illegally modified TEDs (closed TED openings), high fishing densities (multiple capture of individual turtles), and high capture rates in sampling nets (trynets). Data are not currently available to assess the impacts of these factors on sea turtle mortality.

Other trawl fisheries operating in waters under Federal jurisdiction that are known to capture sea turtles include, but are not limited to, summer flounder, calico scallop, blue crab, whelk, cannonball jellyfish, horseshoe crab, and mid-Atlantic directed finfish trawl fisheries and the *Sargassum* fishery. The summer flounder fishery is the only trawl fishery (other than the shrimp fishery) with federally mandated TED use as a result of high mortality rates. In the winter of 1991-1992, Epperly *et al*. (1995) documented a total of 1,063 sea turtle captures in the flounder fishery, and 89 to 191 of the captures were estimated to have been killed. Sea turtle capture rates in other trawl fisheries currently are not available, but are assumed to be relatively small. The harvest of *Sargassum* by trawlers can result in incidental capture of post-hatchlings and habitat destruction (Schwartz 1988, Witherington 2002).

Regulations regarding trawl fisheries under state jurisdiction are highly variable. Some states, including Virginia, Maryland, and Florida, maintain offshore areas permanently closed to trawling. The State of Georgia requires the use of NMFS-approved TEDs in all trawl fisheries operating in state waters. South Carolina uses a water-temperature trigger to ensure whelk trawling occurs when sea turtles are less abundant. With the exception of the shrimp fishery, TEDs are not required in most state-sponsored trawl fisheries.

Longline Fisheries

The principal longline fishery that may impact loggerheads in the neritic environment is the bottom longline fishery for sharks, which operates in summer off the Mid-Atlantic States and all year long off the south Atlantic and Gulf states. NMFS has estimated that 20 sea turtles (assumed to be mostly loggerheads) are killed in that fishery every year (NMFS 2001).

Gillnets

Although a detailed summary of gill net fisheries operating off the Atlantic and Gulf Coasts of the U.S. was presented in NMFS (2001), the dearth of sea turtle mortality data for these fisheries precluded a quantitative analysis of their impact on loggerhead survival. Many states (South Carolina, Georgia, Florida, Louisiana, and Texas) have prohibited gill nets, but there remain active fisheries in other states and in Federal waters. The impact of some of these fisheries, particularly those using large mesh nets, could be catastrophic. In the spring of 2000, the monkfish fishery north of Cape Hatteras, North Carolina, was responsible for approximately 280 sea turtle mortalities over a 2-week period (NMFS, unpublished data).

Poundnets and Weirs

Poundnets are fixed gear composed of a series of poles driven into the bottom upon which netting is suspended. Poundnets basically operate like a trap with the pound being constructed with a series of funnels leading to a bag that is open at the top, and a long (200 to 400 meters) linear "hedge" or leader of netting that extends from shallow to deeper water where the pound is located. Sea turtles trapped in the pound, which is composed of small mesh webbing, are safe from injury and often take advantage of their situation by feeding on the fishes trapped with them. Such turtles may be released easily when the fishermen pull the nets (Mansfield *et al*. 2002). However, sea turtle mortalities have been documented in the hedging of poundnets. Large mesh (greater than 12-inch stretch leaders) may act as a gill net, entangling sea turtles by the head or fore flippers (Bellmund *et al*. 1987). Nets with small mesh hedging usually present no threat to loggerheads (Mansfield *et al*. 2002, Morreale and Standora 1998, Epperly *et al*. 2000). Chesapeake Bay appears to be the primary location where poundnets with large mesh hedging are used (NMFS 2001). In the early 1980s, 3 to 33 percent of all sea turtle mortalities in Virginia were attributed to large mesh leaders in the Bay (Bellmund *et al*. 1987). At that time, 173 such nets were being fished. However, the fishery has declined since then and in 2000 only 20 large mesh nets remained in the Bay (Mansfield *et al*. 2002). Despite close scrutiny of the poundnet fishery in spring 2002 by NMFS and Virginia State authorities, only seven sea turtles could be documented entangled in poundnet leaders out of more than 200 strandings (Virginia Institute of Marine Science, unpublished data).

Hook and Line Fisheries

In 2001, NMFS documented a total of 1,877 commercial vessels engaged in hook and line fisheries targeting snapper-grouper, Gulf reef fish, king mackerel, Spanish mackerel, and sharks in the South Atlantic and Gulf of Mexico (NMFS, unpublished data). The magnitude of sea turtle capture and mortality associated with these fisheries is not known (NMFS, unpublished data). In addition to commercial fisheries, the recreational hook and line fishery is extensive. Turtle captures on hook and line gear are not uncommon, but the level of take and percent mortality are unknown. For hook and line fisheries, it is assumed that most turtles are released alive, although ingested hooks and entanglement in associated monofilament/steel line have been documented as the probable cause of death for some stranded turtles.

Pot/Trap Fisheries

Pots/traps are commonly used in the capture of crabs, lobster, and reef fish. These traps vary in size and configuration but all are attached to a surface float by means of a line leading to the trap. Turtles can become entangled in trap lines below the surface of the water and subsequently drown. In other instances, stranded turtles have been recovered entangled in trap lines with the trap in tow. Loggerhead turtles may be particularly vulnerable to entanglement in trap lines because of their attraction to, or attempts to feed on, species caught in the traps and epibionts (living organisms) growing on traps, trap lines, and floats. Recently a small number of loggerhead entanglements have been recorded in whelk pot bridles in the Middle Atlantic area. Approximately 0.8 percent of stranded sea turtles (1997-2001 average) were found entangled in pot/trap line (NMFS, unpublished data).

Other Fisheries

Incidental captures have been reported from "fish traps" in Massachusetts, Rhode Island, New York, and Florida (NMFS 2001). However, fish traps and poundnets are very similar in construction, and the two terms have been used interchangeably in places. Haul seines and channel nets have been reported to take loggerheads in North Carolina (NMFS 2001), but it is not known how many loggerhead mortalities are caused by these fisheries, if any.

Purse seines are used in the Gulf of Mexico and along the Atlantic coast to capture bluefin tuna and menhaden. The tuna fishery is confined to water over the continental shelf and no sea turtle mortalities have been observed in this fishery (NMFS 2001). The menhaden fishery is pursued close to the coast and in large estuaries such as Chesapeake Bay. Thus, the potential for loggerhead interactions with this fishery may be higher than for the tuna fishery. Although no interactions were observed between sea turtles and purse seines in a study of finfish bycatch in Chesapeake Bay (Herbert Austin, Virginia Institute of Marine Science, personal communication, 2000), sea turtles trapped in menhaden purse seines might be impaled on the grates of inlet pipes used to suck the catch into the hold, and these animals could be mortally injured.

RESOURCE USE (NON-FISHERIES)

Illegal Harvest
Illegal directed harvesting of juvenile and adult loggerhead turtles in the waters of the continental United States and United States Caribbean is uncommon, but no estimates of the level of take exist. During the period 1983 to 1989, the Florida Marine Patrol made three arrests for illegal possession of whole turtles and 25 arrests for illegal possession of turtle parts within Florida.

Oil and Gas Exploration, Development, and Production
Several activities associated with offshore oil and gas production are known to impact loggerhead turtles, including oil spills, water quality (operational discharge), seismic surveys, explosive platform removal, platform lighting, and noise from drillships and production activities. Currently, there are 3,443 federally regulated offshore platforms in the Gulf of Mexico dedicated to the production of natural gas and oil. Additional state-regulated platforms are located in waters under state jurisdiction (Texas and Louisiana). As a result of newly developed floating and subsea production systems, oil production in the Gulf of Mexico is predicted to increase by 160 percent between 1995 and 2006 (Minerals Management Service 2002). There are currently no active leases off the Atlantic coast.

All loggerhead lifestages are vulnerable to the harmful effects of oil through direct contact, degradation of food resources, and loss of habitat (Minerals Management Service 2000). Vargo *et al*. (1986) reported that sea turtles would be at substantial risk if they encountered an oil spill or large amounts of tar in the environment. In a review of available information on debris ingestion, Balazs (1985) reported that tar balls were the second most prevalent type of debris ingested by sea turtles. Exposure to petroleum products can be fatal to all lifestages of loggerhead turtles (Vargo *et al*. 1986). Physiological experiments showed that sea turtles exposed to petroleum products may suffer inflammatory dermatitis, ventilatory disturbance, salt gland dysfunction or failure, red blood cell disturbances, immune response, and digestive disorders (Vargo *et al*. 1986, Lutz and Lutcavage 1989, Lutcavage *et al*. 1995).

Operational discharge of produced waters, drill muds, and drill cuttings are routinely discharged in marine waters as a result of petroleum production activities (Minerals Management Service 2000). Loggerhead turtles may bioaccumulate heavy metals found in drill muds resulting in debilitation or death. Oil exploration and development on live bottom areas may disrupt foraging grounds by smothering benthic organisms with sediments and drilling muds (Coston-Clements and Hoss 1983).

Petroleum seismographic cannons produce intense noise at both high and low frequencies and have the potential to harm sea turtles. The effects of seismic survey activity have not been studied in sea turtles (Lutcavage *et al*. 1997).

The explosive removal of offshore oil and gas platforms is known to have impacts on loggerhead turtles ranging from capillary damage, disorientation, loss of motor control, and mortality. Klima *et al*. (1988) examined the effects of underwater explosions on sea turtles and found five of eight turtles exposed to explosions at distances varying between 229 meters and 915 meters were rendered unconscious. Loggerheads found closer to detonation sites would likely suffer fatal injuries. From 1987 to 2001, NMFS observers reported only one loggerhead mortality and three injured/stunned loggerhead turtles during the removal of approximately 1,300 offshore oil structures in the Gulf of Mexico (NMFS, unpublished data). The small number of sea turtles observed during platform removal may be a result of the inability of observers stationed at the surface to assess the impacts of explosions on submerged sea turtles.

The impacts of offshore lighted oil production platforms on loggerhead turtles are unknown. Lighted platforms may attract hatchlings making them more susceptible to predation (de Silva 1982). Neritic juveniles and adults may be attracted by high prey concentrations around the structures, making them more susceptible to ingestion of petroleum products.

Although turtle hearing is not well studied, drillships and production facilities produce sound at varying frequencies and intensities that may influence turtle behavior (Minerals Management Service 2000). Captive loggerheads have exhibited sound-induced swimming behavior when exposed to low frequency sound (Lenhardt *et al*. 1983). Sound transmissions could increase surfacing behavior and change foraging patterns around production sites.

Power Plant Entrapment

The entrainment and entrapment of loggerhead turtles in saltwater cooling intake systems of coastal power plants has been documented in New Jersey, North Carolina, Florida, and Texas (Roithmayr and Henwood 1982; Ernest *et al*. 1989; S. Manzella, personal communication; T. Henson, personal communication; R. Schoelkopf, personal communication). Average annual incidental capture rates for most coastal plants from which captures have been reported amount to several turtles per plant per year. One notable exception is the St. Lucie Nuclear Power Plant located on Hutchinson Island, Florida. During the first 15 years of operation (1977-1991), an average of 130 loggerheads per year were captured in the intake canal with a mortality rate of 6.6 percent. Over the last 10 years, loggerhead captures have almost doubled (average of 253 per year), while mortality rates have decreased to 0.5 percent per year (Quantum Resources, Inc., unpublished data).

Boat Strikes

Propeller and collision injuries from boats and ships are common in sea turtles. From 1997 to 2001, 12.7 percent of all stranded turtles were documented as having sustained some type of propeller or collision injuries, although it is not known what proportion of these injuries were post or ante-mortem (NMFS, Sea Turtle Stranding and Salvage Network (STSSN), unpublished data). Boat-related injuries are recorded at higher frequencies in areas of high boating traffic. From 1991 to 2000, an average of 22 percent of stranded turtles in Florida showed signs of a boat-related injury (FFWCC, unpublished data).

Military Explosions and Exercises
Military maneuvers involving explosives may potentially harm loggerheads, but specific information on their impacts is not available (National Research Council 1990).

CONSTRUCTION AND DEVELOPMENT

Dredging
Periodic dredging of sediments from navigational channels is necessary to provide for the passage of large commercial and military vessels. The negative impacts of dredging include destruction or degradation of habitat and incidental mortality of sea turtles. Channelization of inshore and nearshore habitat and the subsequent disposal of dredged material in the marine environment can destroy or disrupt resting or foraging grounds (including grass beds and coral reefs) and may affect nesting distribution through the alteration of physical features in the marine environment (Hopkins and Murphy 1980).

The capture and mortality of sea turtles by hopper dredges was identified as a problem in the late 1970s. During a 3-month period in 1980, dredging operations in the Port of Canaveral, Florida, ship channel were responsible for the mortality of at least 71 sea turtles (National Research Council 1990). To minimize mortality associated with hopper dredging, Dickerson *et al*. (1995) conducted a study to determine the seasonal abundance of sea turtles in southeastern ship channels. Loggerhead turtles were rarely captured when water temperatures fell below 16^{o}C. As a result, dredging activities in the south Atlantic were restricted to winter months when turtles are less abundant. Several other methods have been employed to reduce incidental capture and mortality of loggerheads in hopper dredges. The U.S. Army Corps of Engineers funded research to develop a plow-like deflector designed to push or move turtles away from the suction of the draghead (Nelson and Shafer 1996). In addition, shrimp trawlers have been employed to capture and relocate sea turtles prior to or during dredging operations. In 2001, NMFS allowed for the take of 92 loggerheads in shipping channels in the Atlantic and Gulf of Mexico. Documented take levels were substantially lower. Most of the loggerheads taken during dredging operations are juveniles and subadults. Other types of dredges (e.g., clamshell and pipeline) have not been implicated in the incidental take of sea turtles.

Channel Blasting
UNDER DEVELOPMENT

Bridge Blasting
UNDER DEVELOPMENT

Water Diversions
UNDER DEVELOPMENT

Marina and Dock Development

The development of marinas and private or commercial docks in inshore waters can negatively impact turtles through destruction or degradation of foraging habitat. Sanger and Holland (2002) found that docks were not a major source of environmental contamination in South Carolina; however, dock construction was associated with suburban development, which represented a major source of environmental degradation to tidal creeks and associated marsh habitats. Dock proliferation may also result in increased boat and vessel traffic and higher propeller and collision related mortality. Fueling facilities at marinas can result in the discharge of oil, gas, and sewage into sensitive estuarine habitats.

ECOSYSTEM ALTERATIONS

Trophic Changes from Overfishing

Anthropogenic disruptions of natural ecological interactions have been difficult to discern. However, Seney *et al*. (in press) documented a major shift in the diet of juvenile loggerheads in the Chesapeake Bay between 1981 and 2001, apparently caused by overharvest of horseshoe crabs (*Limulus polyphemus*). Lutcavage and Musick (1985) found that horseshoe crabs strongly dominated the diet of juvenile loggerheads in Chesapeake Bay in 1980-1981. Subsequently, horseshoe crabs began to be harvested by several types of gear primarily to be used as bait in the eel and whelk pot fisheries. Atlantic coast horseshoe crab landings increased by an order of magnitude (0.5 to 6.0 million pounds) between 1980 and 1997, and in 1998 the Atlantic States Marine Fisheries Commission had to implement a horseshoe crab fishery management plan to curtail catches (Atlantic States Marine Fisheries Commission 1998).

The decline in horseshoe crab availability has apparently caused a diet shift in juvenile loggerheads, which in recent years have been found to primarily ingest fishes. The presence of fish remains in loggerhead guts has previously been interpreted to be the result of some kind of fishing gear interaction because loggerheads are not considered to be sufficiently agile to capture healthy free swimming fishes (Bellmund *et al*. 1987). Thus, the overharvesting of horseshoe crabs may be leading to more frequent interactions between fishing gear and loggerheads as the turtles attempt to actively feed on fish trapped in poundnets or gillnets, or on discards from these fisheries (Seney *et al*. in press).

Aquaculture

The concern for sea turtles with regard to aquaculture netting is the rigidity of the net and the size of the mesh. The larger the mesh size, the greater the probability of entanglement (though even very small mesh can entangle a turtle, often wrapping around the nail on the rear or foreflippers). The more slack in the net, the greater the probability of entanglement. With regard to entanglement in lines/cables the concerns are identical to marine mammals. Studies have shown that sea turtle hatchlings (at least of certain species and in certain areas) migrate offshore after hatching and become associated (and dependent upon) floating rafts of seaweed (e.g., *Sargassum*) and other flotsam. Net pens and associated aquaculture structures, depending on their siting, may “collect” seaweed rafts or interfere with their natural passive movements and, therefore, may entangle, capture, or disrupt migratory movements of post-hatchling or pelagic-stage sea turtles.

The issue of artificial lighting at aquaculture facilities and its effects on sea turtles should be included in the habitat alteration section. Anthropogenic lighting has been well documented to misorient hatchling turtles during their transit from nest to sea and has also been documented to misorient hatchlings after their entry into the sea. Studies have also shown that nesting female turtles can be deterred from nesting and/or misoriented at the nesting beach by artificial lights. Artificial lighting at aquaculture facilities, depending on their siting, may misorient hatchlings and/or adult females in the proximity of nesting beaches. No studies have been conducted on the effects of artificial lighting offshore of nesting beaches, but such studies must be a component of any thorough investigation into the impacts of aquaculture on sea turtles.

Siting of aquaculture facilities should take into consideration proximity of nesting beaches, foraging habitat, and migratory pathways of sea turtles. Only certain of these are identifiable at this time. Additional research will be needed as aquaculture sites are selected or considered. Vessel strikes are a significant threat to sea turtles in certain areas of high vessel traffic. Increases in vessel traffic that result from aquaculture operations must be evaluated with respect to its effects on resident or migratory sea turtle populations. All of the disease, predation, and alteration of behavior issues/concerns apply to sea turtles as well as marine mammals.

Eutrophication
UNDER DEVELOPMENT

POLLUTION

Marine Debris Entanglement
Loggerhead turtles have been found entangled in a wide variety of materials including steel and monofilament line, synthetic and natural rope, plastic onion sacks, and discarded plastic netting materials (Balazs 1985; Plotkin and Amos 1988; NMFS, unpublished data). From 1997 to 2001, 1.8 percent of stranded sea turtles found on Atlantic and Gulf of Mexico beaches were entangled in fishing gear. Monofilament line appears to be the principal source of entanglement for loggerheads in U.S. waters (1.4 percent; 1997-2001 average), followed by fishing net (0.4 percent; 1997-2001 average) and pot/trap line (0.8 percent; 1997-2001 average). Less than 1 percent of stranded sea turtles in 2000 were found entangled in other marine debris (NMFS, unpublished data). In Florida, the number of stranded sea turtles found entangled in marine debris has remained constant at about 5 percent of the total strandings since 1991 (FFWCC, unpublished data). Records from Florida indicate that some entanglement results from netting and monofilament line that has accumulated on both artificial and natural reefs. These areas are often heavily fished, resulting in snagging of hooks and discarding of lines. Turtles foraging and/or resting in these areas can become entangled and drown (FFWCC, unpublished data). The alignment of persistent marine debris along convergences, rips, and driftlines, and the concentration of young sea turtles along these fronts, increases the likelihood of entanglement at this life history stage (Carr 1987a, Witherington 2002).

Marine Debris Ingestion
Sea turtles have been found to ingest a wide variety of debris items, such as plastic bags, raw plastic pellets, plastic and styrofoam pieces, tar balls, and balloons. Effects of debris ingestion

can include direct obstruction of the gut, absorption of toxic byproducts, and reduced absorption of nutrients across the gut wall (Balazs 1990). Studies conducted by Lutz (1990) revealed that both loggerhead and green turtles actively ingested small pieces of latex and plastic sheeting. Physiological data indicated a possible interference in energy metabolism or gut function, even at low levels of ingestion. Persistence of the material in the gut lasted from a few days to 4 months (Lutz 1990).

Oil Pollution
The deleterious effects of oil pollution on sea turtles have been well documented (Lutcavage *et al*. 1997). Turtles in oil slicks experience prolonged physical contact with floating oil. This contact can cause significant changes in respiration, diving patterns, energy metabolism, and blood chemistry. The turtle salt glands appear to be particularly sensitive to oil pollution. Prolonged salt gland failure interferes with water balance and ion regulation and may be fatal (Lutcavage *et al*. 1997). Annually about 1 percent of all sea turtle strandings along the U.S. east coast have been associated with oil, but higher rates of 3 to 6 percent have been observed in South Florida and Texas (Teas 1994, Rabalais and Rabalais 1980, Plotkin and Amos 1990).

Toxins
Pollution sources other than oil that may affect sea turtles include persistent chlorinated hydrocarbons and heavy metals. Long-lived carnivorous species such as loggerheads would tend to bioaccumulate these compounds (Rybitski *et al*. 1995, Lutcavage *et al*. 1997). However, organochlorine concentrations found in sea turtles have been much lower than those found in marine mammals and birds (George 1997), probably due to the much lower metabolic rates of the turtles. The impacts of these compounds on loggerheads are unknown.

Thermal Pollution (from power plants)
UNDER DEVELOPMENT

Desalinization Plant Pollution
UNDER DEVELOPMENT

SPECIES INTERACTIONS

Predation
Large sharks may prey upon neritic stage loggerheads. Tiger sharks (*Galeocerdo cuvieri*) and bull sharks (*Carcharhinus leucas*) are the species most often reported to contain sea turtle remains (Compagno 1984, Simpfendorfer *et al*. 2001). The magnitude of loggerhead mortality caused by sharks in the western North Atlantic is unknown.

Disease and Parasites
Loggerhead turtles are affected by a variety of health problems, although relatively few diseases have been documented in wild populations. At least two bacterial diseases have been described in wild loggerhead populations, including bacterial encephalitis and ulcerative stomatitis/obstructive rhinitis/pneumonia (George 1997). There are few reports of fungal infections in wild loggerhead populations. Homer *et al*. (2000) documented systemic fungal

infections in stranded loggerheads in Florida. Both bacterial and fungal infections are common in captive sea turtles (Herbst and Jacobson 1995, George 1997).

Viral diseases have not been documented in free-ranging loggerheads, with the possible exception of the sea turtle fibropapilloma disease (STFD), which may have a viral etiology (Herbst and Jacobson 1995, George 1997). Although STFD reaches epidemic proportions in some wild green turtle populations, the prevalence of this disease in most loggerhead populations is thought to be small. An exception is Florida Bay where approximately 11 percent of the loggerhead population is afflicted with STFD (Schroeder *et al*. 1997).

A variety of endoparasites, including gastrointestinal and cardiovascular trematodes, tapeworms, and gastrointestinal nematodes, have been described in loggerheads (Herbst and Jacobson 1995). Heavy infestations of endoparasites may contribute to debilitation and/or mortality in sea turtles. Trematode eggs and adults were seen in a variety of tissues including the spinal cord and brain of debilitated loggerhead turtles during an epizootic in South Florida during 2000. Trematodes have been implicated as a possible cause of the epidemic.

Ectoparasites, including leeches and barnacles, may have debilitating effects on loggerhead turtles. Large marine leech infestations may result in anemia and act as vectors for other disease-producing organisms (George 1997). Barnacles are generally considered to be innocuous although some burrowing species may penetrate the body cavity resulting in mortality (Herbst and Jacobson 1995).

Although many health problems have been described in wild populations through the necropsy of stranded turtles, the significance of diseases on the ecology of wild loggerhead populations is not known (Herbst and Jacobson 1995). Several researchers have initiated health assessments to study health problems in free-ranging turtle populations. Sampling methods for these assessments have included capturing sea turtles with modified shrimp nets (Segars *et al*. 2001) and sampling adult females on nesting beaches (Deem *et al*. 2002). To date, these assessments have focused on establishing normal baseline blood chemistry values and conducting physical exams. As more assays become available, researchers hope to assess the prevalence of infectious diseases in wild loggerhead populations.

Red Tide

A red tide is a higher-than-normal concentration of microscopic algae. In Florida, the species that causes most red tides is *Karenia brevis*, a type of microalgae known as a dinoflagellate (Florida Marine Research Institute 2003). This organism produces a toxin that has been documented as a mortality factor in birds and marine mammals and is a suspected mortality factor in sea turtles. During three red tide events along the west coast of Florida, sea turtle stranding trends indicated that these events were acting as an additional mortality factor. A loggerhead that washed ashore alive during a red tide event displayed symptoms that suggested acute brevitoxicosis (e.g., uncoordinated movements and lethargic but otherwise robust and healthy in appearance) and completely recovered within days of being removed from the area of the red tide. The concentration of brevitoxin in the liver and stomach contents of eight turtles that were found dead during red tides ranged from 10-570 nanograms per gram. These concentrations ranged higher than the concentrations of brevitoxin found in Florida manatees

(*Trichechus manatus latirostris*) that were determined to have died from brevitoxicosis. Concentrations of brevitoxin in the livers and stomach contents of two turtles that did not strand during red tides were less than 0.05-0.29 nanograms per gram (Redlow *et al*. in press).

Although the organism that causes Florida's red tide is found almost exclusively in the Gulf of Mexico, blooms have been found off the east coast of Florida, and a bloom was detected off the coast of North Carolina in 1987. Scientists believe the Florida Current and Gulf Stream Current carried *Karenia brevis* out of the Gulf of Mexico, around south Florida, and up to the Carolina coast. Other types of microorganisms cause different kinds of red tides (now called harmful algal blooms) in other parts of the world as well (Florida Marine Research Institute 2003).

OTHER FACTORS

Climate Change
Ocean warming and climate change can have long-term effects on trophic dynamics by affecting food availability and species interactions.

Natural Catastrophes
Storm events that include high winds and currents may cause post-hatchling turtles to be washed back on nesting beaches (Carr and Meylan 1980). Post-hatchlings, or washback hatchlings, are turtles that have left nesting beaches, spent weeks or months at sea, and are then washed back onto the beach with seaweed during storm events.

Cold Water
Loggerheads are susceptible to cold stunning, a phenomenon in which turtles become incapacitated as a result of rapidly dropping water temperatures (Witherington and Ehrhart 1989, Morreale et al. 1992). As temperatures fall below 8-10^{o}C, turtles may lose their ability to swim and dive, often floating to the surface. It appears to be the rate of cooling that precipitates cold stunning rather than the water temperature itself (Milton and Lutz 1997). Sea turtles that overwinter in inshore waters are most susceptible to cold stunning, because temperature changes are most rapid in shallow water (Witherington and Ehrhart 1989).

G.3. OCEANIC ZONE

RESOURCE USE (FISHERIES)

A major threat to the survival of loggerhead turtles during the oceanic stage is the risk of incidental capture in commercial fisheries. Indirect take in fisheries, whether it is the high seas drift nets, longlines, or other fisheries, is a very serious problem for juvenile turtles (National Research Council 1990, Wetherall et al. 1993, Balazs and Pooley 1994, Witzell 1999, Bolten et al. 2000).

Gillnet (Drift) Fisheries
The bycatch of oceanic juveniles has been well documented for the high seas driftnet fishery (Wetherall et al. 1993).

Longline Fisheries

Incidental take of oceanic-stage loggerheads in the swordfish longline fisheries has recently received a lot of attention (Balazs and Pooley 1994; Bolten *et al*. 1994, 2000; Aguilar *et al*. 1995; Laurent *et al*. 1998). The mean size CCL (+/- standard deviation) for loggerheads captured in the swordfish fishery in the Azores during an experiment conducted in 2000 was 49.8 +/- 6.2 centimeters CCL (n = 224; Bolten *et al*. unpublished data) which is significantly larger ($p < 0.001$, Kolmogorov-Smirnov test, ks = 0.6528) than the mean of the overall oceanic-stage population, 34.5 +/-12.6 centimeters CCL (n = 1692, calculated from Bjorndal *et al*. 2000a). The largest size classes in the oceanic stage are the size classes impacted by the swordfish longline fishery. Earlier studies in Azorean waters documenting swordfish longline captures show similar size classes impacted by that fishery (Bolten *et al*. 1994, Ferreira *et al*. 2001). There are no estimates of mortality of turtles caught and released in the swordfish fishery in the Azores. The demographic consequences relative to population recovery of the increased mortality of these size classes have been discussed (Crouse *et al*. 1987; see also Heppell *et al*. in press and Chaloupka in press).

Similar size classes of loggerheads are impacted by longline fisheries in other regions. In the western Mediterranean the mean size of loggerheads caught in drifting longline fisheries was 47.4 +/- 10.4 centimeters CCL (n = 62) and 45.9 +/- 7.5 centimeters CCL (n = 53) in the eastern Mediterranean (Laurent *et al*. 1998). Witzell (1999) reported a mean size of 55.9 +/- 6.5 centimeters CCL (n = 98) for loggerheads caught in the longline fishery from the western North Atlantic, primarily the Grand Banks, Newfoundland, Canada. In the Pacific, the mean size of loggerheads caught by longlines is 57.7 +/- 11.5 centimeters SCL (n = 163, Balazs and Parker unpublished data).

Results from satellite telemetry with satellite-linked time-depth recorders have demonstrated the potential negative impacts of longline hooking on dive behavior and movement patterns of oceanic juveniles. Following release, hooked turtles have a significantly reduced diving behavior (e.g., shallower dive depths) and their movements appear to be influenced to a greater extent by ocean currents – the turtles are less active swimmers and drift with the current (Bolten, Riewald, and Bjorndal, unpublished data). Researchers in Hawaii report different results for movement patterns for longline hooked turtles (Polavino *et al*. 2000), but see Bolten (in press a) for additional discussion.

There are numerous other fisheries that impact oceanic-stage loggerhead populations, and new ones continue to be developed. For example, the fishery for black scabbard (*Aphanopus carbo*) in Madeira has a significant bycatch of oceanic-stage loggerheads (Dellinger and Encarnacao 2000). This fishery is currently under investigation for development in the Azores.

RESOURCE USE (NON-FISHERIES)

Illegal Harvest

Directed take of very small turtles for food is not common. However, directed take for the souvenir trade in polished shells or whole stuffed turtles, such as the once-popular but now illegal tourist trade in Madeira, Portugal (Brongersma 1982), may still exist in some regions. See Nada (2001).

Boat Strikes
Increased tourism into sensitive sea turtle habitats (e.g., whale watching and sport/recreational fishing) may increase the frequency of boat collisions. There are currently no data available to evaluate this threat.

Oil and Gas Exploration, Development, and Production
Loggerheads can be at a significant risk to both the toxic effects of oil spills and, to a lesser extent, boat collisions during transport.

Military Explosions and Exercises
The effects of low frequency sonar on sea turtles may be significant; however, specific effects have not been documented.

ECOSYSTEM ALTERATIONS

Trophic Changes from Overfishing
Oceanic ecosystems are changing as a result of overfishing and pollution. Selective, and usually intense, harvest of species in fisheries will result in changes to the suite of species interactions in this ecosystem (e.g., predator-prey interactions, trophic dynamics and food webs; see Bjorndal in press a). Changes in trophic dynamics may have a major impact on sea turtles; however, data are lacking to quantify these impacts. Changes to trophic interactions may affect availability of prey for loggerheads, and loggerheads may become prey to a new suite of predators following food-web alterations. Decreasing food resources for loggerheads could result in sub-lethal effects in the form of decreased growth rates and reproductive output (Bjorndal in press a). Such sub-lethal effects will be difficult to discern because our knowledge of rates of food intake and rates of growth is poor (Bjorndal in press a). Similar changes to trophic dynamics in this ecosystem can occur from the toxic effects of pollution.

SPECIES INTERACTIONS & EXOTIC SPECIES

Predation
Oceanic-stage loggerheads in Azorean waters are preyed upon by sharks (e.g., tiger sharks, Santos *et al*., personal communications), killer whales (S. Magalhaes, and P. Afonso, personal communications), and probably by any large carnivorous fish or mammal in this habitat. There are no estimates of predation of loggerheads in this life stage.

Disease
There are no definitive reports of sea turtle fibropapilloma disease for oceanic-stage loggerheads, although samples from Madeira are now being evaluated (Elliott Jacobson, University of Florida, personal communication).

POLLUTION

Oil Pollution and Toxins

Toxic effects of pollution (e.g., oil, gas, heavy metals) can have direct effects on turtles (Lutcavage *et al*. 1997) or can alter the habitat by effecting trophic dynamics. Loggerheads can be at a significant risk to both the toxic effects of oil spills and, to a lesser extent, boat collisions during transport.

Marine Debris Ingestion and Entanglement

The lethal and sub-lethal effects of debris ingestion and entanglement are also major concerns (Balazs 1985, Carr 1987a, McCauley and Bjorndal 1999, Witherington in review b). The open ocean is full of debris, and little loggerheads frequently ingest plastics, tar, styrofoam, and monofilament (Carr 1987a, Witherington in review b). This ingestion as well as entanglement is often lethal. The sublethal effects from marine debris ingestion may also have severe consequences but are difficult to quantify. Laboratory feeding trials have documented that post-hatchling loggerheads were not able to adjust their intakes to counter nutrient dilute diets similar to what turtles would experience when ingesting debris (McCauley and Bjorndal 1998). However, the authors suggest that with increasing size, turtles may be better able to adjust their intakes.

OTHER FACTORS

Climate Change

Ocean warming and climate change can have long-term effects on trophic dynamics by affecting food availability and species interactions as described above.

Natural Catastrophes

Hurricanes can have significant short-term effects on trophic dynamics (see above discussion).

H. Past and Ongoing Conservation Efforts

UNDER DEVELOPMENT

GLOSSARY

Benthic (Demersal) Organisms - organisms on the sea floor in either the neritic zone or oceanic.

Epibenthic organisms –

Epipelagic Organisms – organisms that they occupy the upper 200 meters in the oceanic zone.

Neritic Zone - the inshore marine environment (from the surface to the sea floor) where water depths do not exceed 200 meters. The neritic zone generally includes the continental shelf, but in areas where the continental shelf is very narrow or nonexistent, the neritic zone conventionally extends to areas where water depths are less than 200 meters.

Oceanic Zone - the vast open ocean environment (from the surface to the sea floor) where water depths are greater than 200 meters.

Pelagic Organisms – organisms that occupy the water column, but not the sea floor, in either the neritic zone or oceanic zone.

Terrestrial Zone (supralittoral) - the nesting beach where both oviposition and embryonic development occur.

LITERATURE CITED

Ackerman, R.A. 1980. Physiological and ecological aspects of gas exchange by sea turtle eggs. American Zoologist 20(3):575-583.

Ackerman, R.A. 1997. The nest environment and the embryonic development of sea turtles. Pages 83-106 *in* Lutz, P.L. and J.A. Musick (editors). The Biology of Sea Turtles. CRC Press. Boca Raton, Florida.

Ackerman, R.A., T. Rimkus, and R. Horton. 1991. The hydric structure and climate of natural and renourished sea turtle nesting beaches along the Atlantic coast of Florida. Unpublished report to Florida Department of Natural Resources. 61 pages.

Arianoutsou, M. 1988. Assessing the impacts of human activities on nesting of loggerhead sea turtles (*Caretta caretta* L.) on Zákynthos Island, Western Greece. Environmental Conservation 15(4):327-334.

Austin, D.F. 1978. Exotic plants and their effects in southeastern Florida. Environmental Conservation 5(1):25-34.

Bain, R. S.D. Jewell, J. Schwagerl, and B.S. Neely, Jr. 1997. Sea turtle nesting and reproductive success at the Hobe Sound National Wildlife Refuge (Florida) 1972-1985. U.S. Fish and Wildlife Service unpublished report. 72 pages.

Balazs, G.H. 1985. Impact of ocean debris on marine turtles: entanglement and ingestion. Pages 387-429 i*n* Shomura, R.S. and H.O. Yoshida (editors). Proceedings of the Workshop on the Fate and Impact of Marine Debris, 26-29 November 1984, Honolulu, Hawaii. NOAA Technical Memorandum NMFS/SWFC-54.

Bolten, A.B. In press. Active swimmers - passive drifters: The oceanic juvenile stage of loggerheads in the Atlantic system. *In* Bolten, A.B. and B.E. Witherington (editors). The Biology and Conservation of the Loggerhead Sea Turtle. Smithsonian Institution Press, Washington D.C.

Bolten, A.B. and G.H. Balazs. 1995. Biology of the early pelagic stage - the lost year. Pages 579-581 *in* Bjorndal, K.A. (editor). Biology and Conservation of Sea Turtles, Revised Edition. Smithsonian Institution Press, Washington, D.C.

Bolten, A.B. and B.E. Witherington (editors). *In press*. The Biology and Conservation of the Loggerhead Sea Turtle. Smithsonian Institution Press, Washington D.C.

Bowen, B.W. 1994. Letter dated November 17, 1994, to Sandy MacPherson, National Sea Turtle Coordinator, U.S. Fish and Wildlife Service, Jacksonville, Florida. University of Florida. Gainesville, Florida.

Bowen, B.W. 1995. Letter dated October 26, 1995, to Sandy MacPherson, National Sea Turtle Coordinator, U.S. Fish and Wildlife Service, Jacksonville, Florida. University of Florida. Gainesville, Florida.

Bowen, B., J.C. Avise, J.I. Richardson, A.B. Meylan, D. Margaritoulis, and S.R. Hopkins-Murphy. 1993. Population structure of loggerhead turtles (*Caretta caretta*) in the northwestern Atlantic Ocean and Mediterranean Sea. Conservation Biology 7(4):834-844.

Bustard, H.R. and P. Greenham. 1968. Physical and chemical factors affecting hatching success in the green sea turtle *Chelonia mydas* (L.). Ecology 49:269-276.

Caillouet, C.W., Jr., D.J. Shaver, W.G. Teas, J.M. Nance, D.B. Revera, and A.C. Cannon. 1996. Relationship between sea turtle stranding rates and shrimp fishing intensities in the northwestern Gulf of Mexico: 1986-1989 versus 1990-1993. Fishery Bulletin 94(2):237-249.

Caldwell, D.K. 1959. The loggerhead turtles of Cape Romain, South Carolina. Bulletin of the Florida State Museum of Biological Sciences 4(10):319-348.

Carr, A. 1962. Orientation problems in the high seas travel and terrestrial movements of marine turtles. American Scientist 50(3):358-374.

Carr, A. 1982. Notes on the behavioral ecology of sea turtles. Pages 19-26 *in* Bjorndal, K.A. (editor). Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington, D.C.

Carr, A. 1986. Rips, FADs, and little loggerheads. Bioscience 36:92-100.

Carr, A.F., Jr. 1987a. Impact of nondegradable marine debris on the ecology and survival outlook of sea turtles. Marine Pollution Bulletin 18(6B):352-356.

Carr, A. 1987b. New perspectives on the pelagic stage of sea turtle development. Conservation Biology 1(2):103-120.

Carr, A. and A.B. Meylan. 1980. Evidence of passive migration of green turtle hatchlings in sargassum. Copeia 1980(2):366-368.

Carr, A. and L. Ogren. 1960. The ecology and migrations of sea turtles, 4. The green turtle in the Caribbean Sea. Bulletin of the American Museum of Natural History 121(1):1-48.

Carthy, R.R., A.M. Foley, and Y. Matsuzawa. In press. The incubation environment of nests of the loggerhead turtle: Effects on hatching success and the characteristics of hatchlings. *In* Bolten, A.B. and B.E. Witherington (editors). The Biology and Conservation of the Loggerhead Sea Turtle. Smithsonian Institution Press, Washington D.C.

Christens, E. 1990. Nest emergence lag in loggerhead sea turtles. Journal of Herpetology 24(4):400-402.

Clark, R.R. 1992. Beach conditions in Florida: a statewide inventory and identification of beach erosion problem areas in Florida. Beaches and Shores Technical Memorandum 89-1, 4th Edition. Florida Department of Environmental Protection, Division of Beaches and Shores, Tallahassee, Florida. 208 pages.

Collins, B.T. 1997. ESTEQN: A program to analyze trend data. Canadian Wildlife Service, National Wildlife Research Center, Hull, Province of Quebec, Canada.

Coston-Clements, L. and D.E. Hoss. 1983. Synopsis of data on the impact of habitat alteration on sea turtles around the southeastern United States. NOAA Technical Memorandum NWSSEFC-117. 57 pages.

Cox, J.H., H.F. Percival, and S.V. Colwell. 1994. Impact of vehicular traffic on beach habitat and wildlife at Cape San Blas, Florida. Cooperative Fish and Wildlife Research Unit Technical Report Number 50. 44 pages.

Crain, D.A., A.B. Bolten, and K.A. Bjorndal. 1995. Effects of beach nourishment on sea turtles: review and research initiatives. Restoration Ecology 3(2):95-104.

Crouse, D.T. 1985. Biology and conservation of sea turtles in North Carolina. Unpublished Ph.D. dissertation. University of Wisconsin, Madison, Wisconsin. 216 pages.

Crouse, D.T., L.B. Crowder, and H. Caswell. 1987. A stage-based population model for loggerhead sea turtles and implications for conservation. Ecology 68:1412-1423.

Crowder, L.B., D.T. Crouse, S.S. Heppell, and T.H. Martin. 1994. Predicting the impact of turtle excluder devices on loggerhead sea turtle populations. Ecological Applications 4:437-445.

Crowder, L.B., S.R. Hopkins-Murphy, and J.A. Royle. 1995. The effects of turtle excluder devices (TEDs) on loggerhead sea turtle strandings with implications for conservation. Copeia 1995:773-779.

Daniel, R.S. and K.U. Smith. 1947. The sea-approach behavior of the neonate loggerhead turtle (*Caretta caretta*). Journal of Comparative Physiology and Psychology 40:413-420.

Davis, G.E. and M.C. Whiting. 1977. Loggerhead sea turtle nesting in Everglades National Park, Florida, U.S.A. Herpetologica 33:18-28.

Deem, S.L., L. Starr, T.M. Norton, and W.B. Karesh. In press. Sea turtle health assessment program in the Caribbean and Atlantic. *In* Proceedings of the Twenty-second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum.

Delpech, Y.J. and J.J. Foote. 1998. Effects of three soil cement step-faced revetments on sea turtle nesting habit and hatch success on Casey Key, Florida. Pages 160-163 *in* Epperly, S.P. and J. Braun (compilers). Proceedings of the Seventeenth Annual Sea Turtle Symposium. NOAA Technical Memorandum NMFS-SEFSC-415.

de Silva, G.S. 1995. The status of sea turtle populations in east Malaysia and the South China Sea. Pages 327-337 *in* Bjorndal, K.A. (editor). Biology and Conservation of Sea Turtles, Revised Edition. Smithsonian Institution Press, Washington, D.C.

Dickerson, D.D., K.J. Reine, D.A. Nelson, and C.E. Dickerson. 1995. Assessment of sea turtle abundance in six south Atlantic U. S. channels. U.S. Army Corps of Engineers Waterways Experiment Station. Miscellaneous Paper EL-95-5. 134 pages.

Dodd, C.K., Jr. 1988. Synopsis of the biological data on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758). U.S. Fish and Wildlife Service, Biological Report 88(14). 110 pages.

Dodd, M.G. and A.H. Mackinnon. 1999. Loggerhead turtle (*Caretta caretta*) nesting in Georgia, 1999: implications for management. Georgia Department of Natural Resources unpublished report. 41 pages.

Dodd, M.G. and A.H. Mackinnon. 2000. Loggerhead turtle (*Caretta caretta*) nesting in Georgia, 2000: implications for management. Georgia Department of Natural Resources unpublished report. 47 pages.

Dodd, M.G. and A.H. Mackinnon. 2001. Loggerhead turtle (*Caretta caretta*) nesting in Georgia, 2001. Georgia Department of Natural Resources unpublished report submitted to the U.S. Fish and Wildlife Service for grant E-5-1 “Coastal Endangered Species Management.” 46 pages.

Dutton, P., D. McDonald, and R.H. Boulon. 1994. Tagging and nesting research on leatherback sea turtles (*Dermochelys coriacea*) on Sandy Point, St. Croix, U.S. Virgin Islands, 1994. Annual Report to the U.S. Fish and Wildlife Service. October 1994. 28 pages.

Ecological Associates, Inc. 2002. Habitat conservation plan: a plan for the protection of sea turtles on the beaches of Volusia County, Florida. Plan prepared for Volusia County Environmental Management. 281 pages.

Ehrhart, L.M. 1989. Status report of the loggerhead turtle. Pages 122-139 *in* Ogren, L., F. Berry, K. Bjorndal, H. Kumpf, R. Mast, G. Medina, H. Reichart, and R. Witham (editors). Proceedings of the 2nd Western Atlantic Turtle Symposium. NOAA Technical Memorandum NMFS-SEFC-226.

Ehrhart, L.M. and B.E. Witherington. 1987. Human and natural causes of marine turtle nest and hatchling mortality and their relationship to hatchling production on an important Florida nesting beach. Final Report, Project No. GFC_84_018. Florida Game and Fresh Water Fish

Commission, Nongame Wildlife Program, Technical Report No. 1. Tallahassee, Florida. 141 pages.

Encalada, S.E., K.A. Bjorndal, A.B. Bolten, J.C. Zurita, B. Schroeder, E. Possardt, C.J. Sears, and B.W. Bowen. 1998. Population structure of loggerhead turtle (*Caretta caretta*) nesting colonies in the Atlantic and Mediterranean as inferred from mitochondrial DNA control region sequences. Marine Biology 130:567-575.

Epperly, S.P. 2003. Fisheries-related mortality and turtle excluder devices (TEDs). Pages 339-353 *in* Lutz, P.L., J.A. Musick, and J. Wyneken (editors). The Biology of Sea Turtles, Volume II. CRC Press. Boca Raton, Florida.

Epperly, S.P. and W.G. Teas. 2002. Turtle excluder devices-are the escape openings large enough? Fishery Bulletin 100(3):466-474.

Epperly, S.P., J. Braun, A.J. Chester, F.A. Cross, J.A. Merriner, and P.A. Tester. 1995. Winter distribution of sea turtles in the vicinity of Cape Hatteras and their interactions with the summer flounder trawl fishery. Bulletin of Marine Science 56:547-568.

Ernest, R.G. and R.E. Martin. 1993. Sea turtle protection program performed in support of velocity cap repairs, Florida Power & Light Company St. Lucie Plant. Applied Biology, Inc. Jensen Beach, Florida. 51 pages.

Ernest, R.G. and R.E. Martin. 1999. Martin County beach nourishment project; sea turtle monitoring and studies; 1997 annual report and final assessment. Ecological Associates, Jensen Beach, Florida. 96 pages + tables and figures.

Ernest, R.G., R.E. Martin, N. William-Walls, and J.R. Wilcox. 1989. Population dynamics of sea turtles utilizing shallow coastal waters off Hutchinson Island, Florida. Pages 57-59 i*n* Eckert, S.A., K.L. Eckert, and T.H. Richardson (compilers). Proceedings of the Ninth Annual Workshop on Sea Turtle Conservation and Biology. NOAA Technical Memorandum NMFS-SEFC-232.

Ferris, J.S. 1986. Nest success and the survival and movement of hatchlings of the loggerhead sea turtle (*Caretta caretta*) on Cape Lookout National Seashore. CPSU Technical Report 19. 40 pages.

Florida Department of Environmental Protection, National Oceanic and Atmospheric Administration, and U.S. Department of the Interior. 1997. Damage assessment and restoration plan/environmental assessment for the August 10,1993 Tampa Bay Oil Spill, Volume I - Ecological Injuries - Final. 91 pages.

Florida Marine Research Institute. 2003. What is Red Tide? FMRI web page: http://www.floridamarine.org.

Foley, A.M. 1998. The nesting ecology of the loggerhead turtle (*Caretta caretta*) in the Ten Thousand Islands, Florida. Unpublished Ph.D. dissertation. University of South Florida, Tampa, Florida. 164 pages.

Fritts, T.H. and M.A. McGehee. 1982. Effects of petroleum on the development and survival of marine turtle embryos. U.S. Fish and Wildlife Service report FWS/OBS-82/37. 41 pages.

Geissler, P.H. and J.R. Sauer. 1990. Topics in route-regression analysis. Pages 54-57 *in* J.R. Sauer and S. Droege (editors). Survey Designs and Statistical Methods for the Estimation of Avian Population Trends. U.S. Fish and Wildlife Service, Biological Report 90(1).

George, P.H. 1997. Health problems and diseases in turtles. Pages 363-385 i*n* Lutz, P.L. and J.A. Musick (editors). The Biology of Sea Turtles. CRC Press. Boca Raton, Florida.

Georges, A., C. Limpus, and R. Stoutjesdijk. 1994. Hatchling sex in the marine turtle *Caretta caretta* is determined by proportion of development at a temperature, not daily duration of exposure. Journal of Experimental Zoology 200:432-444.

Gerrodette, T. 1993. Trends: Software for a power analysis of linear regression. Wildlife Society Bulletin 21:515-516.

Glenn, L. 1998. The consequences of human manipulation of the coastal environment on hatchling loggerhead sea turtles (*Caretta caretta*, L.). Pages 58-59 *in* Byles, R. and Y. Fernandez (compilers). Proceedings of the Sixteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-412.

Godfrey, M.H. and N. Mrosovsky. 1997. Estimating the time between hatching of sea turtles and their emergence from the nest. Chelonian Conservation and Biology 2(4):581-585.

Goff, M., M. Salmon, and K.J. Lohmann. 1998. Hatchling sea turtles use surface waves to establish a magnetic compass direction. Animal Behaviour 55:69-77.

Gyuris, E. 1994. The rate of predation by fishes on hatchlings of the green turtle (*Chelonia mydas*). Coral Reefs 13(3):137-144.

Hailman, J.P. and A.M. Elowson. 1992. Ethogram of the nesting female loggerhead (*Caretta caretta*). Herpetologica 48:1-30.

Hanson, J., T. Wibbels, and R.E. Martin. 1998. Predicted female bias in sex ratios of hatchling loggerhead sea turtles from a Florida nesting beach. Canadian Journal of Zoology 76(10):1850-1861.

Hendrickson, J.R. 1958. The green sea turtle *Chelonia mydas* (Linn.) in Malaya and Sarawak. Proceedings of the Zoological Society of London 130:455-535.

Herbst, L.H. and E.R. Jacobson. 1995. Diseases in marine turtles. Pages 593-596 *in* Bjorndal, K.A. (editor). Biology and Conservation of Sea Turtles, Revised Edition. Smithsonian Institution Press. Washington, D.C.

Herren, R.M. 1999. The effect of beach nourishment on loggerhead (*Caretta caretta*) nesting and reproductive success at Sebastian Inlet, Florida. Unpublished Master of Science thesis. University of Central Florida, Orlando, Florida. 138 pages.

Hewavisenthi, S. 1994. Effect of retaining turtle hatchlings in tanks before their release. Pages 56-58 *in* Bjorndal, K.A., A.B. Bolten, D.A. Johnson, and P.J. Eliazar (compilers). Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-351.

Homer, B.L., A. Foley, K.J. Fick, M.C. Lores, A.E. Redlow, and E.R. Jacobson. 2000. Legions, pathogens and toxins identified in 13 stranded marine turtles in Florida. Pages 117-118 i*n* Abreu-Grobois, F.A., R. Briseno-Duenas, R. Marquez, and L. Sarti (compilers). Proceedings of the Eighteenth International Sea Turtle Symposium. NOAA Technical Memorandum NMFS-SEFSC-436.

Hopkins, S.R. and T.M. Murphy. 1980. Reproductive ecology of *Caretta caretta* in South Carolina. South Carolina Wildlife Marine Resources Department Completion Report. 97 pages.

Hopkins-Murphy, S.R., C.P. Hope, and M.E. Hoyle. 1999. A history of research and management of the loggerhead turtle (*Caretta caretta*) on the South Carolina coast. Final Report to the U.S. Fish and Wildlife Service. South Carolina Department of Natural Resources, Charleston, South Carolina. 72 pages.

Hosier, P.E., M. Kochhar, and V. Thayer. 1981. Off-road vehicle and pedestrian track effects on the sea-approach of hatchling loggerhead turtles. Environmental Conservation 8:158-161.

Johnson, S.A., K.A. Bjorndal, and A.B. Bolten. 1996. Effects of organized turtle watches on loggerhead (*Caretta caretta*) nesting behavior and hatchling production in Florida. Conservation Biology 10(2):570-577.

Kaufman, W. and O. Pilkey. 1979. The Beaches are Moving: The Drowning of America's Shoreline. Anchor Press/Doubleday. Garden City, NY. 326 pages.

Kikukawa, A., N. Kamezaki, and H. Ota. 1999. Factors affecting nesting beach selection by loggerhead turtles (*Caretta caretta*): A multiple regression approach. Journal of Zoology 249:447-454.

Klima, E.F., G.R. Gitschlag, and M.L. Renaud. 1988. Impacts of the explosive removal of offshore petroleum platforms on sea turtles and dolphins. Marine Fisheries Review 50:33-42.

Komar, P.D. 1983. Coastal erosion in response to the construction of jetties and breakwaters. Pages 191-204 *in* Komar, P.D. (editor). CRC Handbook of Coastal Processes and Erosion. CRC Press. Boca Raton, Florida.

Kraemer, J.E. 1979. Variation in incubation length of loggerhead sea turtle, *Caretta caretta*, clutches on the Georgia coast. Unpublished Master of Science thesis. University of Georgia, Athens, Georgia. 57 pages.

Kraemer, J.E. and S.H. Bennett. 1981. Utilization of posthatching yolk in loggerhead sea turtles *Caretta caretta*. Copeia 1981(2):406-411.

Kraus, N.C. 1988. The effects of seawalls on the beach: an extended literature review. Journal of Coastal Research Special Issue 4:1-28.

Kraus, N.C. and W.G. McDougal. 1996. The effects of seawalls on the beach: part I, an updated literature review. Journal of Coastal Research 12(3):691-701.

Labisky, R.F., M.A. Mercadante, and W.L. Finger. 1986. Factors affecting reproductive success of sea turtles on Cape Canaveral Air Force Station, Florida, 1985. Final report to the United States Air Force. United States Fish and Wildlife Service Cooperative Fish and Wildlife Research Unit, Agreement Number 14-16-0009-1544, Research Work Order Number 25. 18 pages.

Lenhardt, M.L., S. Bellmund, R.A. Byles, S.W. Harkins, and J.A. Musick. 1983. Marine turtle reception of bone conducted sound. Journal of Auditory Research 23(2):119-126.

Light, P., M. Salmon, and K.J. Lohmann. 1993. Geomagnetic orientation of loggerhead sea turtles: evidence for an inclination compass. Journal of Experimental Biology 182:1-10.

Limpus, C. J. 1971. Sea turtle ocean finding behaviour. Search 2(10):385-387.

Limpus, C.J., V. Baker, and J.D. Miller. 1979. Movement induced mortality of loggerhead eggs. Herpetologica 35(4):335-338.

Limpus, C.J., P. Reed, and J.D. Miller. 1983. Islands and turtles: The influence of choice of nesting beach on sex ratio. Pages 397-402 *in* Baker, J.T., R.M. Carter, P.W. Sammarco, and K.P. Stark (editors). Proceedings of the Inaugural Great Barrier Reef Conference, James Cook University Press, Townsville, Queensland, Australia.

Lohmann, K.J. 1991. Magnetic orientation by hatchling loggerhead sea turtles (*Caretta caretta*). Journal of Experimental Biology 155:37-49.

Lohmann, K.J. and C.M.F. Lohmann. 1994a. Acquisition of magnetic directional preference in hatchling loggerhead sea turtles. Journal of Experimental Biology 190:1-8.

Lohmann, K.J. and C.M.F. Lohmann. 1994b. Detection of magnetic inclination angle by sea turtles: a possible mechanism for determining latitude. Journal of Experimental Biology 194:23-32.

Lohmann, K.J. and C.M.F. Lohmann. 1996. Orientation and open-sea navigation in sea turtles. Journal of Experimental Biology 199:73-81.

Lohmann, K.J. and C.M.F. Lohmann. In press. Orientation mechanisms of hatchling loggerheads. *In* Bolten, A.B. and B.E. Witherington (editors). The Biology and Conservation of the Loggerhead Sea Turtle. Smithsonian Institution Press, Washington D.C.

Lohmann, K.J., J.T. Hester, and C.M.F. Lohmann. 1999. Long-distance navigation in sea turtles. Ethology Ecology & Evolution 11:1-23.

Lohmann, K.J., M. Salmon, and J. Wyneken. 1990. Functional autonomy of land and sea orientation systems in sea turtle hatchlings. Biological Bulletin 179:214-218.

Lohmann, K.J., A.W. Swartz, and C.M.F. Lohmann. 1995. Perception of ocean wave direction by sea turtles. Journal of Experimental Biology 198:1079-1085.

Lohmann, K.J., B.E. Witherington, C.M.F. Lohmann, and M. Salmon. 1997. Orientation, navigation, and natal beach homing in sea turtles. Pages 107-135 *in* Lutz, P.L. and J.A. Musick (editors). The Biology of Sea Turtles. CRC Press. Boca Raton, Florida.

Lutcavage, M.E., P. Plotkin, B. Witherington, and P.L. Lutz. 1997. Human impacts on sea turtle survival. Pages 387-409 *in* Lutz, P.L. and J.A. Musick (editors). The Biology of Sea Turtles. CRC Press. Boca Raton, Florida.

Lutcavage, M.E., P.L. Lutz, G.D. Bossart, and D.M. Hudson. 1995. Physiologic and clinicopathologic effects of crude oil on loggerhead sea turtles. Archives of Environmental Contamination and Toxicology 28:4 7-422.

Lutz, P.L. and M. Lutcavage. 1989. The effects of petroleum on sea turtles: applicability to Kemp's ridley. Pages 52-54 i*n* Caillouet, C.W., Jr., and A.M. Landry, Jr. (compilers). Proceedings of the First International Symposium on Kemp's Ridley Sea Turtle Biology, Conservation and Management. TAMU-SG-89-105.

Mann, T.M. 1977. Impact of developed coastline on nesting and hatchling sea turtles in southeastern Florida. Unpublished Master of Science thesis. Florida Atlantic University, Boca Raton, Florida. 100 pp.

Marcovaldi, M.A., H. Godfrey, and N. Mrosovsky. 1997. Estimating sex ratios of loggerhead turtles in Brazil from pivotal incubation durations. Canadian Journal of Zoology 75:755-770.

Marcus, S.J. and C.G. Maley. 1987. Comparison of sand temperatures between a shaded and unshaded turtle nesting beach in south Florida. (abstract) Seventh Annual Workshop on Sea Turtle Biology and Conservation, February 1987, Wekiva Springs State Park, Florida.

Martin, R.E. 1996. Storm impacts on loggerhead turtle reproductive success. Marine Turtle Newsletter 73:10-12.

McFarlane, R.W. 1963. Disorientation of loggerhead hatchlings by artificial road lighting. Copiea 1:153.

McGehee, M.A. 1990. Effects of moisture on eggs and hatchlings of loggerhead sea turtles (*Caretta caretta*). Herpetologica 46(3):251-258.

Meylan, A., B. Schroeder, and A. Mosier. 1995. Sea turtle nesting activity in the State of Florida 1979-1992. Florida Marine Research Publications Number 52, St. Petersburg, Florida.

Miller, J.D. 1997. Reproduction in sea turtles. Pages 51-81 *in* Lutz, P.L. and J.A. Musick (editors). The Biology of Sea Turtles. CRC Press. Boca Raton, Florida.

Miller, J.D., C.J. Limpus, and M.H. Godfrey. In press. Nest site selection, oviposition, eggs, development, hatching and emergence of loggerhead turtles. *In* Bolten, A.B. and B.E. Witherington (editors). The Biology and Conservation of the Loggerhead Sea Turtle. Smithsonian Institution Press, Washington D.C.

Miller, K., G.C. Packard, and M.J. Packard. 1987. Hydric conditions during incubation influence locomotor performance of hatchling snapping turtles. Journal of Experimental Biology 127:401-412.

Milton, S.L. and P.L. Lutz. 2003. Physiological and genetic responses to environmental stress. Pages 163-197 *in* Lutz, P.L., J.A. Musick, and J. Wyneken (editors). The Biology of Sea Turtles, Volume II. CRC Press. Boca Raton, Florida.

Milton, S.L., S. Leone-Kabler, A.A. Schulman, and P.L. Lutz. 1994. Effects of Hurricane Andrew on sea turtle nesting beaches of South Florida. Bulletin of Marine Science 54(3):974-981.

Minerals Management Service. 2000. Gulf of Mexico deepwater operations and activities - environmental assessment. Minerals Management Service Publication MMS 2000-001. (http://www.gomr.mms.gov/homepg/whatsnew/techann/Deepwater.pdf)

Minerals Management Service. 2002. MMS updates oil & gas production rate projections to 2006; steep rise in oil. Minerals Management Service News Release, June 10, 2002. (http://www.gomr.mms.gov/homepg/whatsnew/newsreal/020610.html)

Moody, K. 1998. The effects of nest relocation on hatching success and emergence success of the loggerhead turtle (*Caretta caretta*) in Florida. Pages 107-108 *in* Byles, R. and Y. Fernandez (compilers). Proceedings of the Sixteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-412.

Moody, K., K. Watson, T. Calleson, R.N. Trindell, and D. Arnold. 2000. Loggerhead turtle (*Caretta caretta*) incubation periods across the Florida Panhandle. Pages 284-286 *in* Kalb, H.J. and T. Wibbels (compilers). Proceedings of the Ninteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-443.

Moran, K.L., K.A. Bjorndal, and A.B. Bolten. 1999. Effects of the thermal environment on the temporal pattern of emergence of hatchling loggerhead turtles *Caretta caretta*. Marine Ecology Progress Series 189:251-261.

Morreale, S.J., A.B. Meylan, S.S. Sadove, and E.A. Standora. 1992. Annual occurrence and winter mortality of marine turtles in New York waters. Journal of Herpetology 26:301.

Mortimer, J.A. 1989. Research needed for management of beach habitat. Pages 236-246 *in* Ogren, L., F. Berry, K. Bjorndal, H. Kumpf, R. Mast, G. Medina, H. Reichart, and R. Witham (editors). Proceedings of the Second Western Atlantic Turtle Symposium. NOAA Technical Memorandum NMFS-SEFC-226.

Mosier, A. 1998. The impact of coastal armoring structures on sea turtle nesting behavior at three beaches on the East Coast of Florida. Unpublished Master of Science thesis. University of South Florida, Tampa, Florida. 111 pages.

Mosier, A.E. and B.E. Witherington. 2002. Documented effects of coastal armoring structures on sea turtle nesting behavior. Pages 304-306 *in* Mosier, A., A. Foley, and B. Brost (compilers). Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFC-477. 369 pages.

Mrosovsky, N. 1968. Nocturnal emergence of hatchling sea turtles: control by thermal inhibition of activity. Nature 220(5174):1338-1339.

Mrosovsky, N. 1980. Thermal biology of sea turtles. American Zoologist 20:531-547.

Mrosovsky, N. 1988. Pivotal temperatures for loggerhead turtles from northern and southern nesting beaches. Canadian Journal of Zoology 66:661-669.

Mrosovsky, N. 1994. Sex ratios of sea turtles. Journal of Experimental Zoology 270(1):16-27.

Mrosovsky, N. and J. Provancha, 1989. Sex ratio of hatchling loggerhead sea turtles: data and estimates from a five year study. Canadian Journal of Zoology 70:530-538.

Mrosovsky, N. and C.L. Yntema. 1980. Temperature dependence of sexual differentiation in sea turtles: implications for conservation practices. Biological Conservation 18:271-280.

Murphy, T.M. 1985. Telemetric monitoring of nesting loggerhead sea turtles subject to disturbance on the beach. Paper Presented at the Fifth Annual Sea Turtle Research Workshop, February 13-16, 1985, Waverly, Georgia.

Murphy, T.M. and S. Hopkins-Murphy. 1990. Homing of translocated gravid loggerhead turtles. Pages 123-124 i*n* Richardson, T.H., J.I. Richardson, and M. Donnelly (compilers). Proceedings of the Tenth Annual Workshop on Sea Turtle Biology and Conservation, NOAA Technical Memorandum NMFS-SEFC -278.

National Marine Fisheries Service. 1987. Sea turtle conservation; shrimp trawling requirements. Federal Register 52(124):24244-24262, June 29, 1987.

National Marine Fisheries Service. 2001. Stock assessments of loggerhead and leatherback sea turtles and an assessment of the impact of the pelagic longline fishery on the loggerhead and leatherback sea turtles of the Western North Atlantic. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-455.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. 1991. Recovery plan for U.S. population of loggerhead turtle (*Caretta caretta*). National Marine Fisheries Service, Washington, D.C.

National Oceanic and Atmospheric Administration and Florida Department of Environmental Protection. 2002. Final damage assessment and restoration plan/environmental assessment for the Fort Lauderdale mystery oil spill - Fort Lauderdale, Florida and vicinity. 83 pages.

National Research Council. 1987. Responding to Changes in Sea Level: Engineering Implications. National Academy Press, Washington, D.C. 148 pages.

National Research Council. 1989. Measuring and Understanding Coastal Processes for Engineering Purposes. National Academy Press, Washington, D.C. 119 pages.

National Research Council. 1990a. Decline of the Sea Turtles: Causes and Prevention. National Academy Press, Washington, D.C. 259 pages.

National Research Council. 1990b. Managing Coastal Erosion. National Academy Press, Washington, D.C. 182 pages.

National Research Council. 1995. Beach Nourishment and Protection. National Academy Press, Washington, D.C. 334 pages.

Nelson, D.A. and B. Blihovde. 1998. Nesting sea turtle response to beach scarps. Page 113 *in* Byles, R. and Y. Fernandez (compilers). Proceedings of the Sixteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-412.

Nelson, D.A. and D.J. Shafer. 1996. Effectiveness of a sea turtle-deflecting hopper dredge draghead in Port Canaveral Entrance Channel, Florida. Miscellaneous Paper D-96-3, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi. 44 pages.

Nelson, D.A., K. Mauck, and J. Fletemeyer. 1987. Physical effects of beach nourishment on sea turtle nesting, Delray Beach, Florida. Technical Report EL-87-15, U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, Mississippi. 56 pages + appendix.

Nelson, K., R. Trindell, B. Witherington, and B. Morford. 2002. An analysis of reported disorientation events in the State of Florida. Pages 323-324 *in* Mosier, A., A. Foley, and B. Brost (compilers). Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-477.

Packard, M.J. and G.C. Packard. 1986. Effect of water balance on growth and calcium mobilization of embryonic painted turtles (*Chrysemys picta*). Physiological Zoology 59(4):398-405.

Packard, G.C., M.J. Packard, and T.J. Boardman. 1984. Influence of hydration of the environment on the pattern of nitrogen excretion by embryonic snapping turtles (*Chelydra serpentina*). Journal of Experimental Biology 108:195-204.

Packard, G.C., M.J. Packard, and W.H.N. Gutzke. 1985. Influence of hydration of the environment on eggs and embryos of the terrestrial turtle *Terrapene ornata*. Physiological Zoology 58(5):564-575.

Packard, G.C., M.J. Packard, T.J. Boardman, and M.D. Ashen. 1981. Possible adaptive value of water exchange in flexible-shelled eggs of turtles. Science 213:471-473.

Packard, G.C., M.J. Packard, K. Miller, and T.J. Boardman. 1988. Effects of temperature and moisture during incubation on carcass composition of hatchling snapping turtles (*Chelydra serpentina*). Journal of Comparative Physiology B 158:117-125.

Parkinson, R.W., L. Lucas, and J.P. Magron. 1999. Physical attributes of nesting beaches: marine turtle monitoring program. Sebastian Inlet Tax District Report. Indiatlantic, Florida. June 16, 1999. 31 pages.

Parmenter, C.J. 1980. Incubation of the eggs of the green sea turtle, *Chelonia mydas*, in Torres Strait, Australia: the effect of movement on hatchability. Australian Wildlife Research 7:487-491.

Pearce, A.F. 2001. Contrasting population structure of the loggerhead turtle (*Caretta caretta*) using mitochondrial and nuclear DNA markers. M.S. thesis. University of Florida, Gainesville, Florida.

Peters, A. and K.J.F. Verhoeven, 1994. Impact of artificial lighting on the seaward orientation of hatchling loggerhead turtles. Journal of Herpetology 28(1):112-114.

Philibosian, R. 1976. Disorientation of hawksbill turtle hatchlings, *Eretmochelys imbricata*, by stadium lights. Copeia 1976(4):824.

Pilcher, N.J. and S. Enderby. 2001. Effects of prolonged retention in hatcheries on green turtle (*Chelonia mydas*) hatchling swimming speed and survival. Journal of Herpetology 35(4):633-638.

Pilkey, O.H. and H.L. Wright III. 1988. Seawalls versus beaches. Journal of Coastal Research Special Issue 4:41-64.

Pilkey, Jr., O.H., D.C. Sharma, H.R. Wanless, L.J. Doyle, O.H. Pilkey, Sr., W. J. Neal, and B.L. Gruver. 1984. Living with the East Florida Shore. Duke University Press, Durham, North Carolina. 55 pages.

Plotkin, P. and A.F. Amos. 1988. Entanglement in and ingestion of marine debris by sea turtles stranded along the south Texas coast. Pages 79-82 i*n* Schroeder, B.A. (compiler). Proceedings of the Eighth Annual Workshop on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFC-214.

Provancha, J.A. and L.M. Ehrhart. 1987. Sea turtle nesting trends at Kennedy Space Center and Cape Canaveral Air Force Station, Florida, and relationships with factors influencing nest site selection. Pages 33-44 *in* Witzell, W.N. (editor). Ecology of East Florida Sea Turtles: Proceedings of the Cape Canaveral, Florida Sea Turtle Workshop. NOAA Technical Report NMFS-53.

Reardon, R. and K. Mansfield. 1997. Annual report-1997 season: Dry Tortugas National Park sea turtle monitoring program, Monroe County, Florida. National Park Service unpublished report. 37 pages.

Redlow, T., A. Foley, and K. Singel. In press. Sea turtle mortality associated with red tide events in Florida. *In* Proceedings of the Twenty-first Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum.

Roithmayr, C. and T. Henwood. 1982. Incidental catch and mortality report. Unpublished National Marine Fisheries Service Report. 20 pages.

Ross, J.P. 1982. Historical decline of loggerhead, ridley, and leatherback sea turtles. Pages 189-195 *in* Bjorndal, K.A. (editor). Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington, D.C.

Routa, R.A. 1968. Sea turtle nest survey of Hutchinson Island, Florida. Quarterly Journal of the Florida Academy of Sciences 30(4):287-294.

Royle, J.A. and L.B. Crowder. 1998. Estimation of a TED effect from loggerhead strandings in South Carolina and Georgia strandings data from 1980-97. Unpublished report, U.S. Fish and Wildlife Service, Laurel, Maryland. 11 pages.

Rumbold, D.G., P.W. Davis, and C. Perretta. 2001. Estimating the effect of beach nourishment on *Caretta caretta* (loggerhead sea turtle) nesting. Restoration Ecology 9(3):304-310.

Ryder, C.E. 1993. The effect of beach renourishment on sea turtles nesting and hatching success at Sebastian Inlet State Recreation Area, East-Central, Florida. Unpublished Master of Science thesis. Virginia Polytechnic Institute and State University, Blacksburg, Virginia. 109 pages.

Salmon, M. and K.J. Lohmann. 1989. Orientation cues used by hatchling loggerhead sea turtles (*Caretta caretta* L.) during their offshore migration. Ethology 83:215-228.

Salmon, M. and J. Wyneken. 1987. Orientation and swimming behavior of hatchling loggerhead turtles *Caretta caretta* L. during their offshore migration. Journal of Experimental Marine Biology and Ecology 109:137-153.

Salmon, M., J. Wyneken, E. Fritz, and M. Lucas. 1992. Seafinding by hatchling sea turtles: role of brightness, silhouette and beach slope as orientation cues. Behaviour 122 (1-2):56-77.

Sanger, D.M. and A.F. Holland. 2002. Evaluation of impacts of dock structures on South Carolina estuarine environments. South Carolina Department of Natural Resources, Marine Resources Research Institute Technical Report 99. 82 pages.

Schmelz, G.W. and R.R. Mezich. 1988. A preliminary investigation of the potential impact of Australian pines on the nesting activities of the loggerhead turtle. Pages 63-66 *in* Schroeder, B.A. (compiler). Proceedings of the Eighth Annual Workshop on Sea Turtle Conservation and Biology. NOAA Technical Memorandum NMFS-SEFSC-214.

Schroeder, B.A. 1981. Predation and nest success in two species of marine turtles (*Caretta caretta* and *Chelonia mydas*) at Merritt Island, Florida. Florida Scientist 44(1):35.

Schroeder, B.A. 1987. Strandings of marine turtles along the United States Atlantic and Gulf of Mexico coasts (1986). Page 19 i*n* Serino, J.L. (compiler). Proceedings of the Seventh Annual Workshop on Sea Turtle Biology and Conservation, February 1987, Wekiwa Springs State Park, Florida.

Schroeder, B.A. and A.E. Mosier. 2000. Between a rock and a hard place: coastal armoring and marine turtle nesting habitat in Florida. Pages 290-292 *in* Abreu-Grobois, F.A., R. Briseño-Dueñas, R. Márquez, and L. Sarti (compilers). Proceedings of the Eighteenth Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-436.

Schwartz, F.J. 1988. Aggregation of young hatchling sea turtles in *Sargassum* off North Carolina. Marine Turtle Newsletter 42:9-10.

Segars, A., D. Whitaker, M., Arendt, S.R. Hopkins-Murphy, and B.W. Stender. In press. Analyses of blood chemistry in trawl-caught *Caretta caretta* and *Lepidochelys kempii* in the southeastern USA. *In* Proceedings of the Twenty-first Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum.

Shoop, C.R., C.A. Ruckdeschel, and R.D. Kenney. 1999. Long-term trends in size of stranded juvenile loggerhead sea turtles (*Caretta caretta*). Chelonian Conservation and Biology 3(3):501-504.

Sobel, D. 2002. A photographic documentation of aborted nesting attempts due to lounge chairs. Page 311 *in* Mosier, A., A. Foley, and B. Brost (compilers). Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-477.

Spotila, J.R., E.A. Standora, S.J. Morreale, G.J. Ruiz, and C. Puccia. 1983. Methodology for the study of temperature related phenomena affecting sea turtle eggs. U.S. Fish and Wildlife Service Endangered Species Report 11.

Stancyk, S.E. 1995. Non-human predators of sea turtles and their control. Pages 139-152 *in* Bjorndal, K.A. (editor). Biology and Conservation of Sea Turtles, Revised Edition. Smithsonian Institution Press. Washington, D.C.

Stancyk, S.E., O.R. Talbert, and J.M. Dean. 1980. Nesting activity of the loggerhead turtle *Caretta caretta* in South Carolina, II: protection of nests from raccoon predation by transplantation. Biological Conservation 18:289-298.

Steinitz, M.J., M. Salmon, and J. Wyneken. 1998. Beach renourishment and loggerhead turtle reproduction: a seven year study at Jupiter Island, Florida. Journal of Coastal Research 14(3):1000-1013.

Tait, J.F. and G.B. Griggs. 1990. Beach response to the presence of a seawall. A comparison of field observations. Shore and Beach 58(2):11-28.

Talbert, O.R., Jr., S.E. Stancyk, J.M. Dean, and J.M. Will. 1980. Nesting activity of the loggerhead turtle *(Caretta caretta)* in South Carolina I: a rookery in transition. Copeia 1980(4):709-718.

Trindell, R., D. Arnold, K. Moody, and B. Morford. 1998. Post-construction marine turtle nesting monitoring results on nourished beaches. Pages 77-92 *in* Tait, L.S. (compiler). Proceedings of the 1998 Annual National Conference on Beach Preservation Technology. Florida Shore & Beach Preservation Association, Tallahassee, Florida.

Turtle Expert Working Group. 1998. An assessment of the Kemp's ridley (*Lepidochelys kempii*) and loggerhead (*Caretta caretta*) sea turtle populations in the western North Atlantic. NOAA Technical Memorandum NMFS-SEFSC-409.

Turtle Expert Working Group. 2000. Assessment update for the Kemp's ridley and loggerhead sea turtle populations in the western North Atlantic. NOAA Technical Memorandum NMFS-SEFSC-444.

U.S. Fish and Wildlife Service. 1978. Listing and protecting loggerhead sea turtles as "threatened species" and populations of green and olive ridley sea turtles as threatened species or "endangered species." Federal Register 43(146):32800-32811, July 28, 1978.

Vargo, S., P. Lutz, D. Odell, E. VanVleet, and G. Bossart. 1986. Study of the effects of oil on marine turtles: final report. OCS Study MMS 86-0070. U.S. Department of the Interior, Mineral Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana. 3 vols. 360 pp.

Wang, J.H., J.K. Jackson, and K.J. Lohmann. 1998. Perception of wave surge motion by hatchling sea turtles. Journal of Experimental Marine Biology and Ecology 229:177-186.

Witherington, B.E. 1986. Human and natural causes of marine turtle clutch and hatchling mortality and their relationship to hatchling production on an important Florida nesting beach. Unpublished Master of Science thesis. University of Central Florida, Orlando, Florida. 141 pages.

Witherington, B.E. 1991. Orientation of hatchling loggerhead turtles at sea off artificially lighting and dark beaches. Journal of Experimental Marine Biology and Ecology 149(1991):1-11.

Witherington, B.E. 1992. Behavioral responses of nesting sea turtles to artificial lighting. Herpetologica 48(1):31-39.

Witherington, B.E. 1994. Some lost-year turtles found. Pages 194-197 *in* Schroeder, B.A. and B.E. Witherington (compilers). Proceedings of the Thirteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-341.

Witherington, B.E. 1995. Observations of hatchling loggerhead turtles during the first few days of the lost year(s). Pages 154-157 *in* Richardson, J.I. and T.H. Richardson (compilers). Proceedings of the Twelfth Annual Sea Turtle Workshop on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-361.

Witherington, B.E. 1997. The problem of photopollution for sea turtles and other nocturnal animals. Pages 303-328 *in* Clemmons, J.R. and R. Buchholz (editors). Behavioral Approaches to Conservation in the Wild. Cambridge University Press, Cambridge, United Kingdom.

Witherington, B.E. 1999. Reducing threats to nesting habitat. Pages 179-183 *in* Eckert, K.L., K.A. Bjorndal, F.A. Abreu-Grobois, and M. Donnelly (editors). Research and Management Techniques for the Conservation of Sea Turtles. IUCN/SSC Marine Turtle Specialist Group Publication No. 4.

Witherington, B.E. 2002. Ecology of neonate loggerhead turtles inhabiting lines of downwelling near a Gulf Stream front. Marine Biology 140:843-853.

Witherington, B.E. and K.A. Bjorndal. 1991. Influences of artificial lighting on the seaward orientation of hatchling loggerhead turtles (*Caretta caretta*). Biological Conservation 55:139-149.

Witherington, B.E. and L.M. Ehrhart. 1989. Hypothermic stunning and mortality of marine turtles in the Indian River Lagoon system, Florida. Copeia 1989:696.

Witherington, B.E. and R.E. Martin. 2000. Understanding, assessing, and resolving light-pollution problems on sea turtle nesting beaches. 2nd edition, revised. Florida Marine Research Institute Technical Report TR-2. 73 pages.

Witherington, B.E. and M. Salmon. 1992. Predation on loggerhead turtle hatchlings after entering the sea. Journal of Herpetology 26(2):226-228.

Witherington, B., K. Bjorndal, and A. Bolten. 1994. An evaluation of the use of dune structure to reduce effects of artificial lighting on hatchling sea turtle sea-finding and offshore orientation. Unpublished Final Report to the U.S. Air Force, Cape Canaveral Air Force Station, and the Florida Cooperative Fish and Wildlife Research Unit, University of Florida, under Research Work Order 75. 38 pages.

Witherington, B.E., K.A. Bjorndal, and C.M. McCabe. 1990. Temporal pattern of nocturnal emergence of loggerhead turtle hatchlings from natural nests. Copeia 1990(4):1165-1168.

Witherington, B., C. Crady, and L. Bolen. 1996. A "hatchling orientation index" for assessing orientation disruption from artificial lighting. Pages 344-347 *in* Keinath, J.A., D.E. Barnard, J.A. Musick, and B.A. Bell (compilers). Proceedings of the Fifteenth Annual Symposium on the Biology and Conservation of Sea Turtles. NOAA Technical Memorandum NMFS-SEFSC-387.

Witherington, B., L. Lucas, and C. Koeppel. In press. Nesting sea turtles respond to the effects of ocean inlets. *In* Proceedings of the Twenty-first Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum.

Wood, D.W. and K.A. Bjorndal. 2000. Relation of temperature, moisture, salinity, and slope to nest site selection in loggerhead sea turtles. Copeia 2000(1):119-128.

Wyneken, J. and M. Salmon. 1992. Frenzy and postfrenzy swimming activity in loggerhead, green and leatherback hatchling sea turtles. Copeia 1992(2):478-484.

Wyneken J. and M. Salmon. 1996. Aquatic predation, fish densities, and potential threats to sea turtle hatchlings from open-beach hatcheries: final report. Technical Report 96-04, Florida Atlantic University, Boca Raton, Florida.

Wyneken, J., M. Salmon, and K.J. Lohmann. 1990. Orientation by hatchling loggerhead sea turtles *Caretta caretta L*. in a wave tank. Journal of Experimental Marine Biology and Ecology 139:43-50.

Wyneken, J., L. DeCarlo, L. Glenn, M. Salmon, D. Davidson, S. Weege, and L. Fisher. 1998. On the consequences of timing, location and fish for hatchlings leaving open beach hatcheries. Pages 58-59 *in* Byles, R., and Y. Fernandez (compilers). Proceedings of the Sixteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-412.